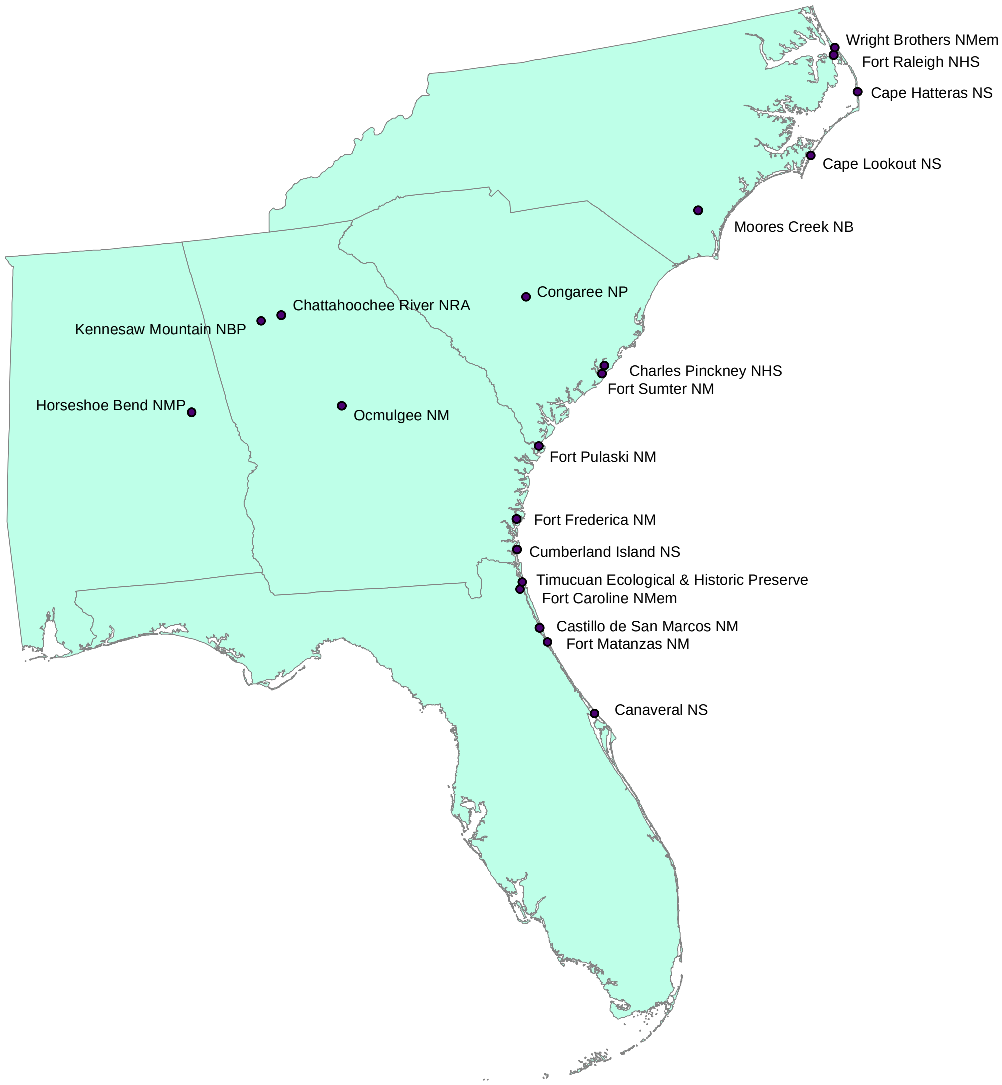
National Park Service
U.S. Department of the Interior

Southeast Coast Network
Atlanta, Georgia

Appendix 7 – Conceptual Models of Southeast Coast Network Ecosystems



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Revision History

Date	Description
15 August 2004	Submitted with final version of the Phase I Report.
22 February 2005	Revised based on comments from national Inventory & Monitoring program office. Ecosystems discussed expanded from four to six; nested models for each system developed or incorporated into report.
18 April 2005	Introductory material revised and citations added to *Literature Cited* section.
07 August 2006	Draft Phase III Report Version

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Introduction & Approach

Development of conceptual models is a required step in design of the Vital Signs Monitoring Program for each network. This requirement is based on lessons learned about monitoring program design from the NPS experience with its prototype parks program, and from many other monitoring programs. What these lessons demonstrate is that every monitoring effort is based on some underlying understanding of how the ecosystem in question works. This underlying understanding forms a mental model, often not written for others to read and discuss. To ensure a successful monitoring effort, these underlying models need to be explicit and available for discussion, evaluation, and refinement (Maddox et al. 1999).

Models are purposeful representations of reality (Starfield et al. 1994). Conceptual models provide a mental picture of how something works, with the purpose of communicating that explanation to others. Models (of all types) work best when they include only the minimum amount of information needed to meet the model's purpose (Starfield 1997).

Conceptual models play several useful roles in monitoring program design, including:

- Formalizing current understanding of the context and scope of the ecological processes important in the area of interest;
- Expanding our consideration across traditional discipline boundaries, fostering integration of biotic and abiotic information;
- Facilitating communication among scientists from different disciplines, between scientists and managers, and between managers and the public.

The key point about conceptual models is their role in communication among people with different points of view (Abel et al. 1998). Conceptual models can take a variety of forms—from narrative descriptions to schematic diagrams or flowcharts with boxes and arrows. Regardless of form, the success of a model depends on its ability to share viewpoints and develop a common understanding based on multiple viewpoints.

Within the National Park Service Inventory & Monitoring program, the development of conceptual models has the specific purpose of guiding the process of selecting *Vital Signs*—information-rich attributes that will be selected for long-term monitoring. With this purpose, a critical role of the conceptual models discussed below is to identify (a) key resources and functions, (b) natural and anthropogenic agents of change, and (c) expected ecosystem responses within Southeast Coast Network ecosystems. With the drivers of change identified, the types of ecological changes most important for park managers to detect can be evaluated. Knowing what changes it is desired to detect is the foundation for the selection of vital signs.

This appendix is organized in three sections, each with increasing levels of detail. In the first section of an overall theoretical framework is outlined to guide the development of system- and issue-specific conceptual models, and to provide insights applicable to the prioritization and selection of vital signs. This framework consists of two components: 1) a simple, general model describing factors and processes controlling the structure, function, and sustainability of ecosystems, and 2) a set of corollary hypotheses concerning key aspects of ecosystem dynamics with particular implications for vital-signs monitoring.

In the second section, a set of detailed conceptual models is introduced to describe hypotheses concerning the mechanisms by which agents of change (both natural and anthropogenic) affect SECN ecosystems. In the third section of the chapter, these ecosystem-based conceptual models are supplemented with narrative descriptions (and relevant citations) of key model components. Pathway models specific to individual vital signs are not included in this Appendix, but will be incorporated into protocols as they are developed.

This appendix (as well as Chapter II) remains under development, and sections described above are in various stages of completion. Conceptual models will continue to be developed and revised during Fall 2006 following peer review, consultation with subject-matter experts, and on-going literature review.

Theoretical Framework and Key Concepts of Ecosystem Dynamics

Rationale

Ecosystems, including the full suite of abiotic and biotic components and processes that they encompass, are fundamental resources of SECN parks. A premise of the SECN vital-signs monitoring program is that the many species and landscapes valued by NPS staff, visitors, and society at large cannot be conserved in the absence of an ecosystem focus. This perspective is based on practical as well as theoretical considerations. Walker (1995) noted that "Given our inadequate understanding and knowledge of how many and which kinds of species occur in an ecosystem, the best way to approach the problem of conserving them all is to ensure that the system continues to have the same overall structure and function"—a practical view shared by many conservation biologists (e.g., Noss 1990, Franklin 1993, Noon et al. 1999). Contemporary ecological theory further suggests that conservation should emphasize the maintenance of *ecosystem processes* because ecosystems and ecosystem components are inherently dynamic both in space and in time and thus cannot be conserved as static entities (Pickett et al. 1992, Christensen et al. 1996). The process-based perspective described for ecosystems is equally important to other levels of organization including populations, species, and landscapes. Ecosystems are connected with other ecosystems by flows of materials, energy, and organisms in spatially structured landscape mosaics (Turner et al. 2001). Thus landscape-level considerations are encompassed in the ecosystem approach of the SECN.

Ecosystems and landscapes can be represented conceptually in many different ways along continua of complexity and specificity. Conceptual models are “caricatures of nature” (Holling et al. 2002) designed to describe and communicate ideas about how nature works. For purposes of vital signs monitoring, it is useful to begin with a simple, general model that summarizes ideas about ecosystem sustainability. This general model and a set of corollary hypotheses provide a theoretical framework for aspects of the monitoring plan related to ecosystem structure and function.

Interactive Controls of Ecosystem Sustainability

Jenny (1941, 1980) proposed that soil and ecosystem processes are determined by five *state factors* – climate, organisms, relief (topography), parent material, and time since disturbance. Jenny’s state-factor approach has been widely applied as a framework for examining temporal and spatial variations in ecosystem structure and function (e.g., Walker and Chapin 1987, Vitousek 1994, Seastedt 2001). Chapin and colleagues (1996) recently extended this framework to develop a set of ecological principles concerning ecosystem sustainability. They defined “...a sustainable ecosystem as one that, over the normal cycle of disturbance events, maintains its characteristic diversity of major functional groups, productivity, and rates of biogeochemical cycling” (Chapin et al. 1996). These ecosystem characteristics are determined by a set of four “interactive controls”–climate, soil-resource supply, major functional groups of organisms, and disturbance regime–and these interactive controls both govern and respond to ecosystem attributes (Figure A7-1). Interactive controls are constrained by the five state factors, which determine the “constraints of place” (Dale et al. 2000).

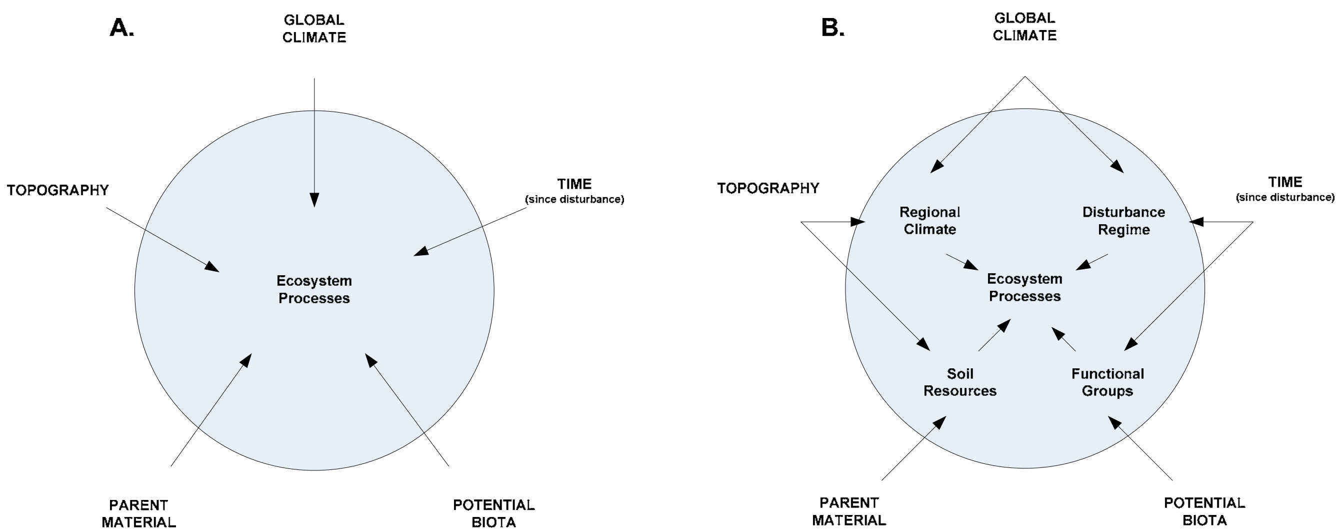

Figure A7-1. Relationship (A) between Jenny’s (1941) state factors and ecosystem processes, and (B) among state factors, interactive controls, and ecosystem processes. The circle represents the boundary of the ecosystem (from Chapin et al. 1996).

By substituting water quality and quantity for soil resources in the model, the interactive-control model can be applied to aquatic as well as terrestrial ecosystems (Chapin et al. 1996). This extends the utility of the model, and it suggests further clarifications. Soil, water, and air are the media from which primary producers acquire resources. As the abiotic matrix that supports the biota, they form the foundation of ecosystems. These media also are characterized by condition attributes (e.g., temperature, stability) that affect the physiological performance of organisms.

Water and air qualities are accepted concepts with legislative standards. No legislative standards exist for the comparable concept of soil quality, and the concept itself was defined only recently. Karlen and colleagues (1997) defined *soil quality* as “the capacity of a specific kind of soil to function, within natural or managed ecosystem boundaries, to sustain plant and animal productivity, maintain or enhance water and air quality, and support human health and habitation.” Soil quality can be regarded as having 1) an inherent component defined by the soil’s inherent soil properties as determined by Jenny’s (1941) five factors of soil formation, and 2) a dynamic component defined by the change in soil function that is influenced by human management of the soil (Seybold et al. 1999).

In terms of the interactive-control model, the concepts of *water quality* and *soil quality* will be used interchangeably with the more descriptive concepts of *water resources and conditions* and *soil resources and conditions*, respectively.

With respect to climate as it is represented in the interactive-control model, the broader concept of *atmospheric resources and conditions* is more precise, encompassing climatic conditions such as temperature, resources such as precipitation and CO_2, and stressors such as airborne pollutants. This is an important clarification in the context of global environmental changes. Figure A7-2 illustrates the modified version of the interactive-control model, the array of stressors affecting SECN parks, and the first-order pathways linking stressors to SECN ecosystems. Complex, higher order effects occur as the four major controls interact via ecosystem processes.

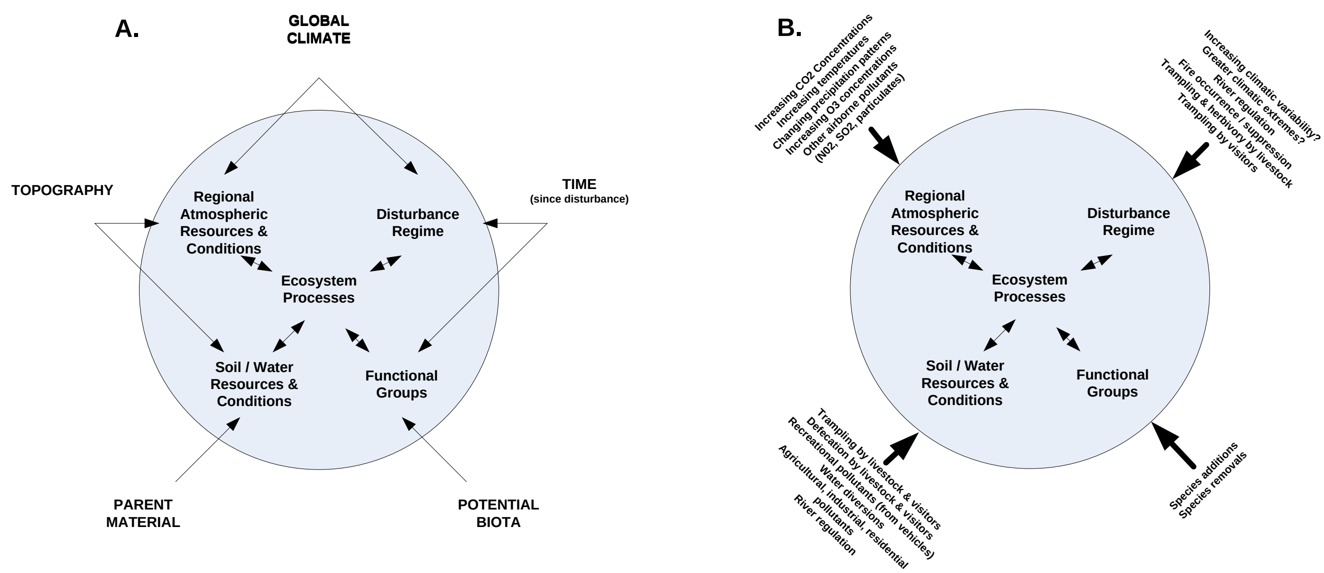

Figure A7-2. Modified version (A) of the interactive-control model that serves as the general ecosystem model for the Southeast Coast Network (SECN), and (B) the array of stressors affecting SECN ecosystems arranged in the model in relation to their first-order effects.

For vital signs monitoring, a key aspect of the interactive-control model is the associated hypothesis that interactive controls must be conserved for an ecosystem to be sustained. Large changes in any of the four interactive controls are predicted to result in a new ecosystem with different characteristics than the original system (Chapin et al. 1996). For example, major changes in soil resources (e.g., through erosion, salinization, fertilization, or other mechanisms) can greatly affect productivity, recruitment opportunities, and competitive relations of plants, and thus can result in major changes in the structure and function of plant communities and higher trophic levels. Changes in vegetation structure can affect the ecosystem’s disturbance regime (e.g., through altered fuel characteristics). These factors and processes in combination can result in a fundamentally different type of ecosystem. Under some circumstances, effects of land uses such as grazing even can affect regional atmospheric resources and conditions through alterations of vegetation and soil conditions that alter ecosystem-atmosphere exchanges of water and energy (e.g., Bryant et al. 1990, Eastman et al. 2001). Additions or losses of species with traits that have strong effects on ecosystem processes also can result in an ecosystem with fundamentally different characteristics – potentially affecting the persistence of previous ecosystem components. Species that affect soil-resource regimes, disturbance regimes, or functional-group structure are those most likely to have profound effects on ecosystem characteristics following their introduction or loss from a system (Vitousek 1990, Chapin et al. 1997). Examples with particular relevance to vital signs monitoring include invasive exotic species that alter ecosystem disturbance regimes (D'Antonio and Vitousek 1992, Mack and D'Antonio 1998) and/or ecosystem resource regimes (Vitousek et al. 1987, Simons and Seastedt 1999). In these cases, the first-order ecosystem change was the introduction of a new functional group.

Ecological Stability, Thresholds, and Resilience

The general ecosystem model describes factors and processes affecting ecosystem structure, function, and sustainability. The notion of ecosystem sustainability is associated with the broader concept of ecological stability. Grimm and Wissel (1997) found 163 definitions of 70 concepts related to ecological stability in the literature. They reduced this chaos to six stability properties:

1. Constancy – staying essentially unchanged.
2. Resilience – returning to the reference state (or dynamic) after a temporary disturbance.
3. Persistence – persistence through time of an ecological system.
4. Resistance – staying essentially unchanged despite the presence of disturbances.
5. Elasticity – speed of return to the reference state (or dynamic) after a temporary disturbance.
6. Domain of attraction – the whole of states from which the reference state (or dynamic) can be reached again after a temporary disturbance.

Sustainable ecosystems, as defined by Chapin and colleagues (1996), are persistent. Inherent in the notions of ecosystem sustainability and persistence is the hypothesis that ecosystems can be caused to cross thresholds and switch from one state (or dynamic) to an alternative state (or dynamic). Of greatest concern from a conservation perspective are alternative states characterized by irreversibly degraded ecosystem structure and function. Ecosystems that have been driven across thresholds of degradation cannot be restored to previous conditions simply by removing the stressor. Costly, manipulative restoration efforts are required (Hobbs and Norton 1996, Whisenant 1999). The success of such restoration efforts usually is uncertain. Arid-land ecosystems and aquatic ecosystems are the most-frequently-cited examples of systems characterized by multiple alternative states (Rapport et al. 1999). Questions concerning ecological thresholds and multiple ecosystem states are at the forefront of theoretical ecology, but threshold issues are not only theoretical. Because of the conceptual link between thresholds and ecosystem sustainability, threshold issues are fundamental to many current questions in applied ecology (e.g., Davenport et al. 1998, Archer and Stokes 2000), including ecosystem management, assessment, and monitoring (Dale et al. 1998, Paine et al. 1998, Andreasen et al. 2001, Herrick et al. 2002, Whitford 2002).

The ball-and-cup heuristic provides a useful means of describing concepts of ecosystem stability (Figure A7-3). In this scheme, the ball represents the ecosystem and the cup represents the domain (or basin) of attraction of the ecosystem. The basin of attraction is analogous to the natural range of variation in the ecosystem. The likelihood of a stochastic perturbation driving the ecosystem across a threshold into a basin of attraction of another state depends on characteristics of the perturbation as well as on the shape of the basin (Scheffer et al. 2001). *Resilience* in Figure A7-3 refers to the size and shape of the basin of attraction, and these correspond to the maximum perturbation that can be absorbed by the ecosystem without resulting in a shift to an alternative state. This definition of resilience follows that of Holling (1973) and encompasses *both* the properties of resilience and resistance as described above by Grimm and Wissel (1997). The ball-and-cup model presented in Figure A7-3 can also be referred to as a *stability landscape* (Scheffer et al. 2001).

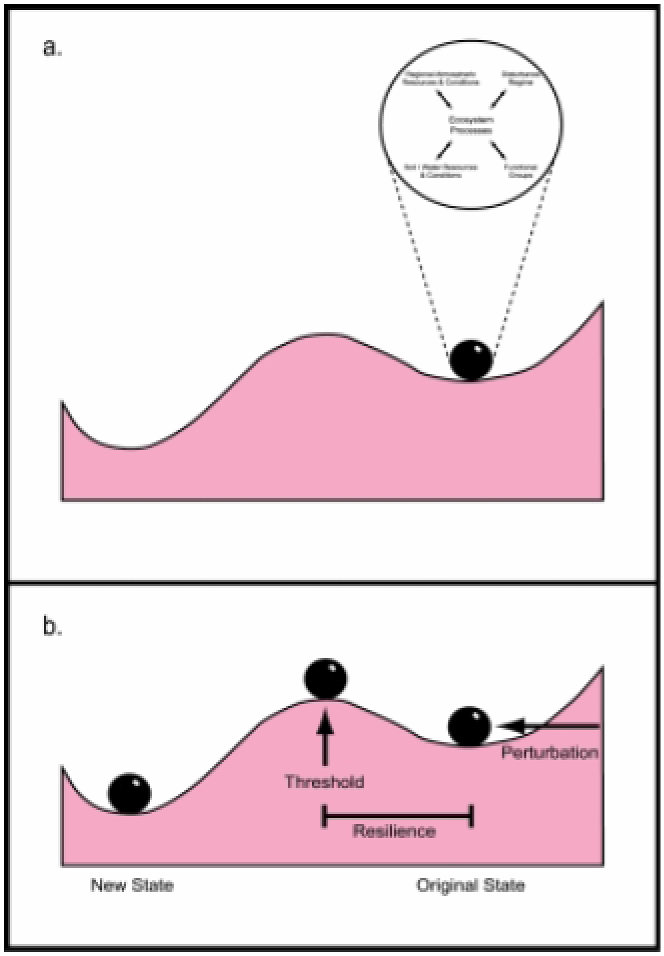

Figure A7-3. Integration (a) of the general ecosystem model with the ball-and-cup heuristic. In (b), *resilience* is illustrated as the magnitude of perturbation that the system can absorb without crossing a threshold from the original dynamic state to a new dynamic state.

Stability landscapes describing the resilience of a particular ecosystem to a particular stochastic perturbation are not static but can be altered as a consequence of gradual changes in environmental conditions that affect ecosystem attributes (Scheffer et al. 2001). Configuration of the stability landscape can vary in relation to environmental conditions. Stated alternatively, ecosystem resilience to the perturbation is condition dependent. For example, fluctuating or gradually changing climatic conditions may alter the stability landscape of an ecosystem and increase the likelihood that a land-use perturbation is capable of driving the ecosystem across a threshold (Tausch et al. 1993). Conversely, accumulated stresses attributable to past and on-going land-use activities may alter the stability landscape of an ecosystem and reduce its resilience to unpredictable (but expected) climatic episodes. Ecosystems can be driven across thresholds abruptly (catastrophically) as resilience erodes under the influence of incremental, cumulative stresses. In arid and semiarid lands, catastrophic shifts among ecosystem states often are preconditioned by impacts of landuse activities on soils and plant-soil interactions (Schlesinger et al. 1990, van de Koppel et al. 1997, Whitford 2002).

Scheffer et al. (2001) noted that the notion of ecological resilience has important implications for resource management:

"Efforts to reduce the risk of unwanted state shifts should address the gradual changes that affect resilience rather than merely control disturbance. Stability domains typically depend on slowly changing variables such as land use, nutrient stocks, soil properties and biomass of long-lived organisms. *These factors may be predicted, monitored, and modified.* In contrast, stochastic events that trigger state shifts (such as hurricanes, droughts or disease outbreaks) are usually difficult to predict or control. Therefore, building and maintaining resilience of desired ecosystem states is likely to be the most pragmatic and effective way to manage ecosystems in the face of increasing environmental change."

System Specific Models

Six dominant ecosystem types have been identified in the Southeast Coast Network: upland forests, riparian / bottomland hardwood forests, rivers & streams, salt marshes & coastal wetlands, barrier islands, and estuaries / nearshore marine systems (Figure A7-4).

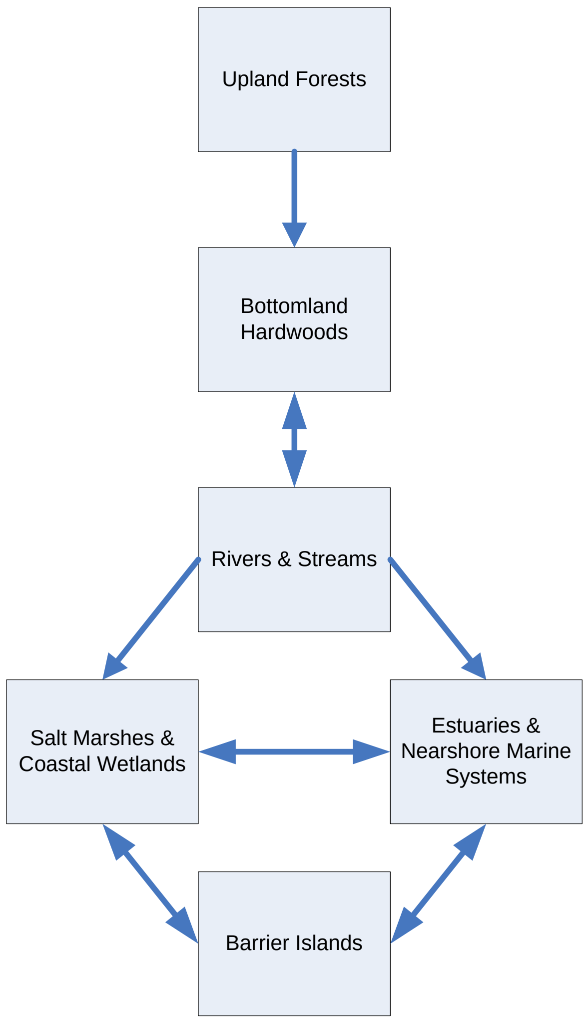

Figure A7-4. The six primary ecosystems within Southeast Coast Network Parks. Arrows indicate how processes and resources in each ecosystem relate to other ecosystems within the Network (i.e., processes or changes within rivers and streams can potentially drive processes or changes within salt marshes or estuarine systems).

Each ecosystem below includes several "layers" of models as follow:

1. In all cases, we have identified or developed an ecosystem-specific generalized model that includes the primary biotic and abiotic resources found within the ecosystem, an their relationships to one another.
2. Where necessary, particularly in terrestrial ecosystems, we have expanded those models to include processes (natural and anthropogenic) that drive successional shifts among dominant community types.
3. Where appropriate we have attempted to also document any major environmental gradients that are known or suspected to explain trends among the parks within the network (or the region)
4. Each ecosystem then contains an ecosystem dynamics model that depicts the major agents of change, interactions among those agents, and the resources that they affect.

Each system is also discussed in both a regional and a network or park-specific context. For further explanatory information on components of each model, a literature review on the agents of change and park resources included within the models are contained in the section following the models (*Ecosystem Components and Key Agents of Change*).

Pineland and Upland forests

The pinelands of the Coastal Plain once extended from the James River in southeastern Virginia to the Trinity River in eastern Texas and covered 24 to 35 million hectares (Frost et al. 1986, Stout and Marion 1993). Longleaf pine savanna was the most common community--the trees, which were thinly distributed, flat-topped, and had limbless lower trunks, occurred in a sea of grasses and diverse wildflowers and carnivorous plants. The historical distribution of pineland communities was determined by moisture supply and fire (Frost et al. 1986). Pines were dominant in habitats ranging from pine flatwoods and mesic savannas to the longleaf pine-turkey oak association in the dry Carolina Sandhills. Longleaf pine was the leading species, with slash pine increasing southward. Both species are now outnumbered by loblolly pine because of fire suppression, conversion to farmland, and commercial timber production (Ware et al. 1993).

The most widespread of the pineland communities, the longleaf pine savanna, occurred widely on the moisture gradient from wet areas and mesic savannas to the dry sandhills and turkey oak associations. The vast, parklike longleaf pine savanna had an herbaceous layer dominated by wiregrass in the southeastern states and by bluestems in Louisiana and eastern Texas. At small scales (1-100 square meters), this herb layer is one of the most diverse in the world; 40 to 75 species of vascular plants have been reported for a single 1-square-meter quadrat and 130 for a 0.1-hectare plot (Clewell 1989). Today, only 14% of the expansive longleaf pine forest remains, with just 3% surviving as old-growth habitat, a loss comparable with or exceeding that of many of the other unique communities in North America (Noss 1989). The dry longleaf pine-turkey oak stands of the sandhills are the most poorly protected areas of this endangered ecosystem (Stout and Marion 1993).

Species that inhabit longleaf pinelands exhibit a high incidence of rarity and endemism. The longleaf pine-wire-grass community includes 191 species of rare plants. Pine communities on the Atlantic Coastal Plain are more diverse and contain a greater number of rare plants. The southeastern pineland community harbors large numbers of federally listed species: 18 plants, 4 reptiles, 4 birds, and 1 mammal, as well as 100 candidates for federal listing (Noss et al. 1995). In addition, the pinelands serve as a major corridor for a large number of migratory birds that winter in the West Indies and South America (Stout and Marion 1993), and they support 170 species of reptiles and amphibians (Dodd, Jr. 1995b). High percentages of these reptile and amphibian species are imperiled (endangered, threatened, or declining): 22% of the salamanders, 15% of the frogs, 34% of the turtles, 31% of the lizards, and 19% of the snakes fall in this category (Dodd, Jr. 1995a).

Lightning fires, occurring at about 1- to 3-year intervals throughout the area, were carried over large areas by wire-grass and pine duff and were stopped only by excessive moisture or abrupt changes in topography. Historically, 10%-30% of the southeastern pinelands burned each year (Ferry et al. 1995); these frequent fires reduced litter accumulation and invasion by competing woody species. Pine seedlings and many of the grasses and forbs present in longleaf pine communities are shade-intolerant, and many require bare mineral soil and reduced competition for germination and early growth. Longleaf pine has several adaptations to minimize fire injury and a large annual needle cast that provides good fuel for future fires (Stout and Marion 1993). The reduction of litter accumulation is essential for the survival of small, rare herbaceous species such as the unique Venus flytrap.

By the time European explorers and settlers arrived in this region, Native Americans had already been augmenting the natural lightning-caused fire regime with annual burning. Set in fall and winter, these fires were used to drive game and improve browse. Early settlers also used fire in winter to improve forage for their livestock, which roamed freely in the forested land.

The longleaf pine forest remained largely intact until the mid-seventeenth century, when the Naval stores industry (that is, products such as turpentine or pitch, originally used to caulk the seams of wooden ships) started to develop in Virginia and then reached its full development in North Carolina in the mid-eighteenth century. Demand then turned to timberland, and despite warnings from late nineteenth-century foresters concerned with regeneration, much of the old-growth forest was cut by the 1920's (Ware et al. 1993).

With much of the timberland being converted to agriculture and much of the wiregrass understory disturbed and fragmented by logging roads and fields, the era of unrestricted ground fires ended. In the absence of fire, other species of pines and woody plants invaded, shading out the regenerating longleaf pine and the sun-loving herbaceous layer. The introduction of livestock also contributed to the end of regeneration by longleaf pine; the nonresinous, carbohydrate-rich meristems of longleaf pine seedlings became favorite livestock forage. In the mesic regions along the coast, extensive areas of longleaf pine were cut, drained, and converted to commercial pine plantations. Finally, the initiation of government-sponsored fire suppression in the 1920's completed the demise of fire-maintained longleaf pinelands in all but a few locations. By 1946 the range of longleaf pinelands had decreased to one-sixth of their former extent, and today only 14% of the original total remains (Frost et al. 1986).

Much of the remaining 2 million hectares of longleaf pine are fragmented and located near developed areas. Winter burning can actually promote woody invasion of the wiregrass understories, but summer burning (the natural fire regime) is considered hazardous near human property. Prescribed burning relies on firebreaks and roads, which further fragment the herbaceous understory and alter local hydrology (Noss 1989). Even though some rare native species respond to other types of disturbance, fire is the most universally important disturbance (Hardin and White 1989).

Of the animals dependent on longleaf pinelands, the best known is the red-cockaded woodpecker, a federally listed species unique for its use of live old-growth or mature second-growth pine trees for cavity excavation (Costa and Walker 1995). The red-cockaded woodpecker is the prime cavity builder in an environment largely free of snags and natural cavities. This species has declined with the loss of longleaf pine habitat; however, intensive management has stabilized several populations (Costa and Walker 1995). Bachman's sparrow, federally listed as threatened, nests in the wire-grass tussocks. The fox squirrel is dependent on the longleaf pine for forage in late summer (Ware et al. 1993). The gopher tortoise, a species whose populations have declined by 80% in the past 100 years (Auffenberg and Franz 1982), is a keystone species in longleaf pine savannahs--more than 300 species of invertebrates and 65 species of vertebrates use burrows dug by gopher tortoises, the only species that creates this microhabitat (Dodd, Jr. 1995a). Recent regional trends are not available for this species. A study in Florida showed that gopher tortoise populations had increased on one study site, decreased on another, and remained stable on three others (data from 1987 to 1988 compared with 1978 to 1979; Mccoy and Mushinsky 1992).

Nearly all parks have upland forest communities, though those community types vary widely across the Network. Natural systems within the network are marked by high levels of plant diversity, and more often than not historical dependence on fire as significant landscape-level drivers of ecosystem function. The elimination of natural burning processes, combined with a long history of silviculture has resulted in forests (regionally) that are highly modified in nature (Figure A7-5).

The current status of upland forested ecosystems is largely driven by climatic drivers, and external processes linked to often rapidly changing landscape dynamics. Successful management of these lands and interpretation of monitoring data collected from them must be performed in the context not only of current, but historical land use practices (grazing, burning, etc.). The cumulative effects of these off-site drivers results in large influxes of invasive species (plant and animal) that interact with native communities (Figure A7-6).

Figure A7-5. (PLACEHOLDER IMAGE / Model FOR NOW). Relationship among plant community types as a function of historic land uses within longleaf pine ecosystems.

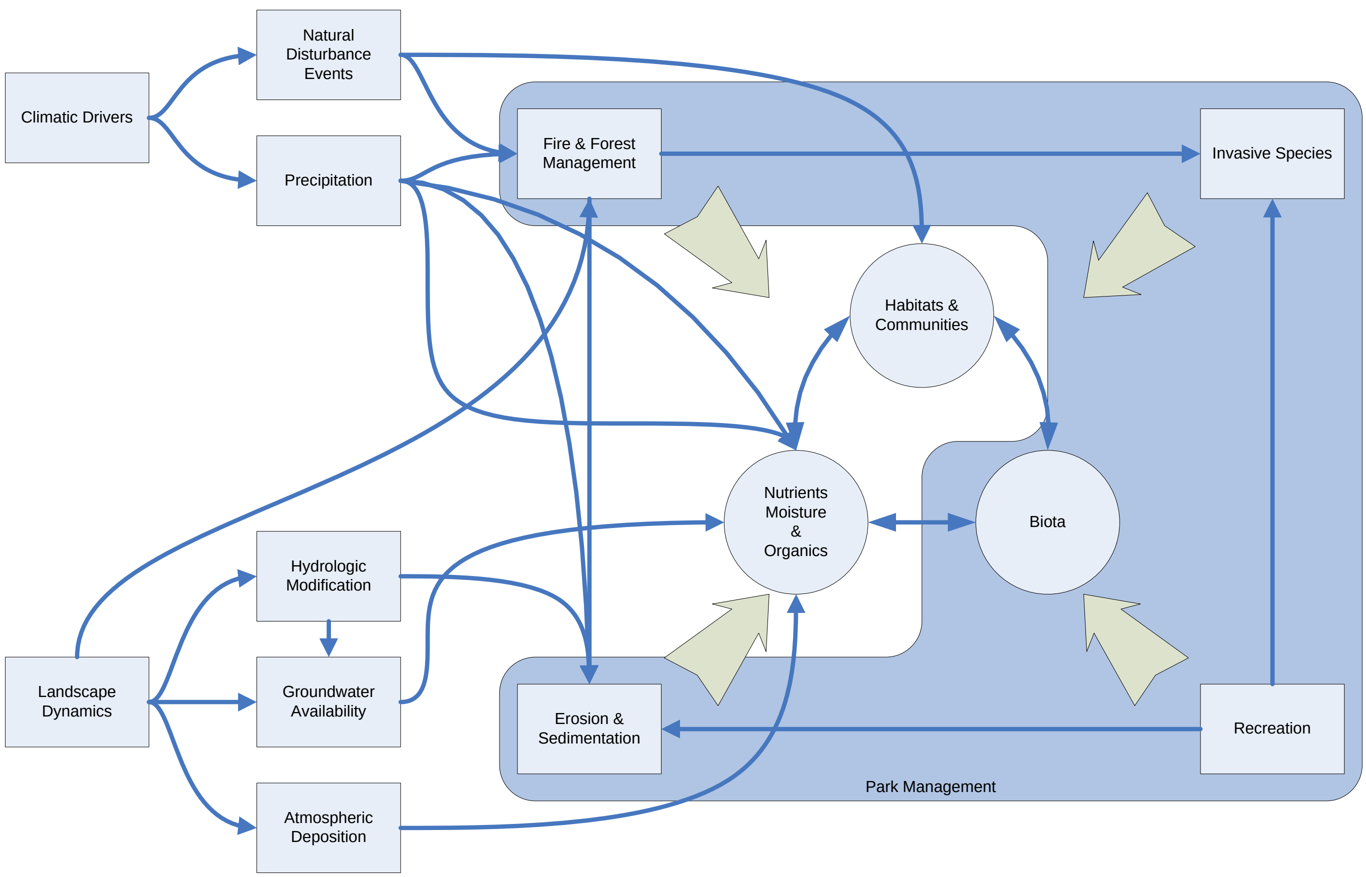

Figure A7-6. Conceptual model of ecosystem dynamics in upland and pineland forests of Southeast Coast Network Parks. Rectangles indicate predominant agents of change; circles represent major components of the ecosystem (detailed in supporting text). Large greenish arrows link agents of change to the entire ecosystem, including biotic, chemical, and physical components. The dark blue area includes those agents of change or resources that are currently / actively managed by NPS.

Bottomland & Floodplain Forests

The Southeast contains 36% of all wetlands and 65% of the forested wetlands of the conterminous United States, even though it makes up only 16% of this area (Keeland et al. 1995). Noss et al. (1995) estimated that 78% of southeastern wetlands were lost between settlement and 1980.

The forested wetlands of the Coastal Plain and Piedmont and the continental interior include bottomland hardwood forests and deepwater alluvial swamps (Sharitz and Mitsch 1993); twelve major forest types have been recognized.

The vegetation of these forests varies in composition and structure according to flooding duration (Larson et al. 1981).

Harris (1989) listed characteristics of these ecosystems that are beneficial to wildlife: hard mast production and a phenology (that is, periodic biological phenomena, such as flowering and breeding, in relation to climate) that is not synchronous with surrounding upland communities, frequent cavity trees, high abundance and biomass of invertebrate wildlife, and a linear distribution throughout the landscape that aids local and regional movement of animals. The seasonal flooding of these habitats makes them less suitable for agriculture; thus, in agricultural landscapes, they are often the only forest refuges available for many mammals, birds, and other species. Bottomland forests were and are very important to many birds in the Southeast, and the extinction of one species, the Carolina parakeet, and the extirpation of another, the ivory-billed woodpecker, are partially the result of fragmentation of this habitat.

Southern floodplain forests may have the largest remaining area of any riparian habitat in the United States (Klopatek et al. 1979, Keeland et al. 1995). Estimates of extent vary widely, however, from 6,600,000 hectares (Klopatek et al. 1979) to 13,000,000 hectares (Abernathy and Turner 1987). This areal extent is decreasing (0.51% per year from 1954 to 1974; Harris and Gosselink 1990), with a total loss of about 63% (Klopatek et al. 1979) to 78% (Noss et al. 1995). These forests have been converted to farmland, industrial parks, and urban areas. Surviving stands are influenced by levee construction, channelization, agricultural runoff, cattle grazing, timber extraction, and invasions of nonindigenous species. Restoration has been attempted, with 65,000 hectares of bottomland forest replanted since 1985, but it is too early to tell if these efforts will be successful (Keeland et al. 1995).

Species and population losses accompany these trends in habitat loss. For example, in Louisiana, Burdick et al. (1989) showed that the number of forest bird species was 15% lower and the number of individual birds 33% lower on transects with 26% forest cover compared with those areas that had 46% forest cover.

Bottomland hardwood forests are one of the dominant riparian ecosystem types in the United States (Mitsch and Gosselink 1993); Congaree National Park contains the largest contiguous tract of old-growth bottomland floodplain forest in the nation. These wetlands represent a transition zone between terrestrial and aquatic ecosystems (Figure A7-7). Ecosystem processes and distributions of both plant and animals are driven at least in part by gradients of flooding frequency, duration and timing (Figure A7-8 and Figure A7-9). Like upland forest communities, bottomland hardwood and riparian forests within network parks are influenced by exotic invasive species, historical land use, and (to a lesser extent) visitor uses. However, bottomland hardwood forests and riparian zones within the network are much more sensitive to landscape dynamics within their watersheds that alter hydrology or water quality (Figure A7-10).

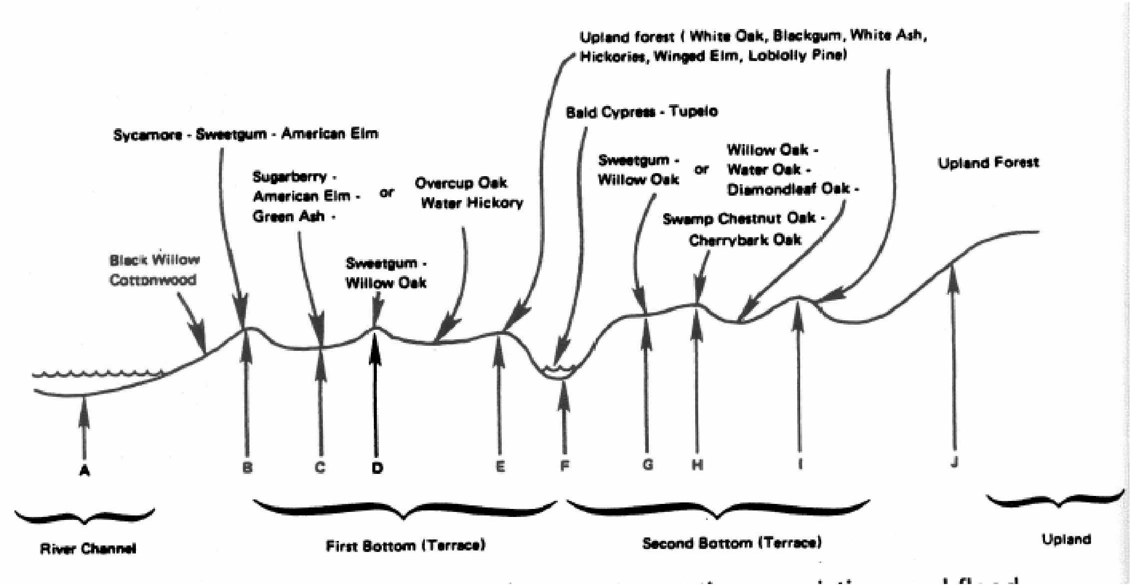

Figure A7-7. General relation between vegetation communities and floodplain microtopography in southeastern floodplains. [A – river channel; B – natural levee; C – backswamp or first terrace; D – low first terrace; E – high first terrace; F – oxbow; G – second terrace flats; H – low second terrace ridge; I – high second terrace ridge; J = upland] (Wharton et al. 1982).

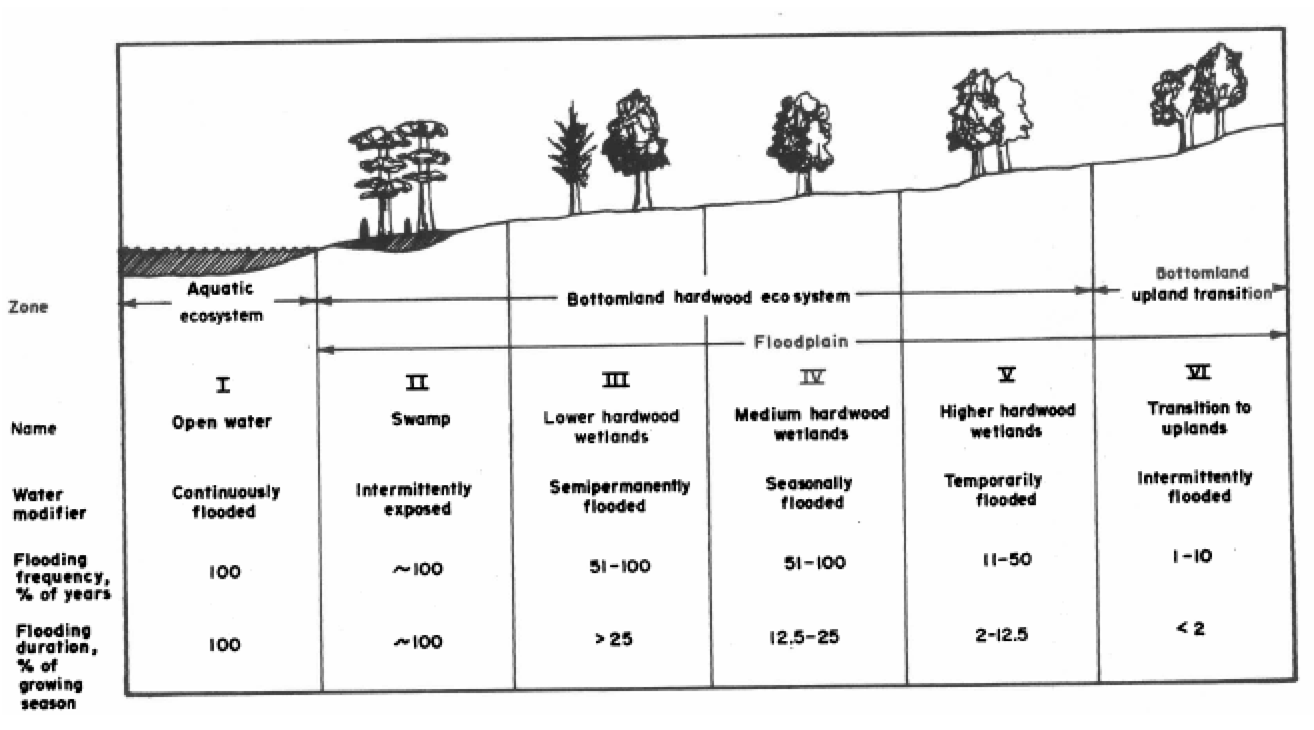

Figure A7-8. Zonal classification of southeastern United States bottomland forest wetlands showing average hydrologic conditions (Mitsch and Gosselink 1993).

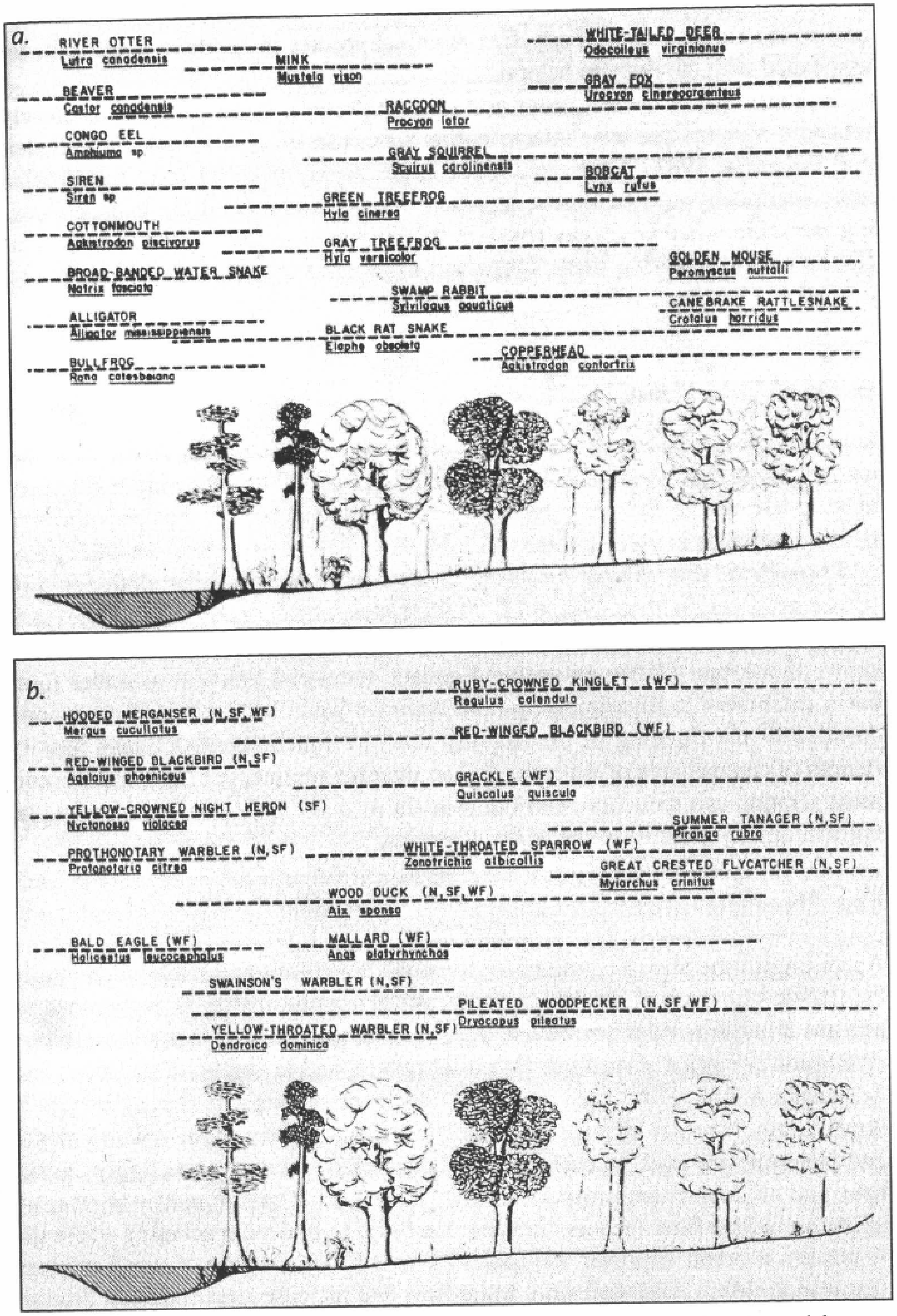

Figure A7-9. Distribution of animals in southern bottomland hardwood forests; *a.* amphibians, reptiles, and mammals, and, *b.* birds in relation to nesting (N), summer foraging (SF), and winter foraging (WF) (Fredrickson 1979).

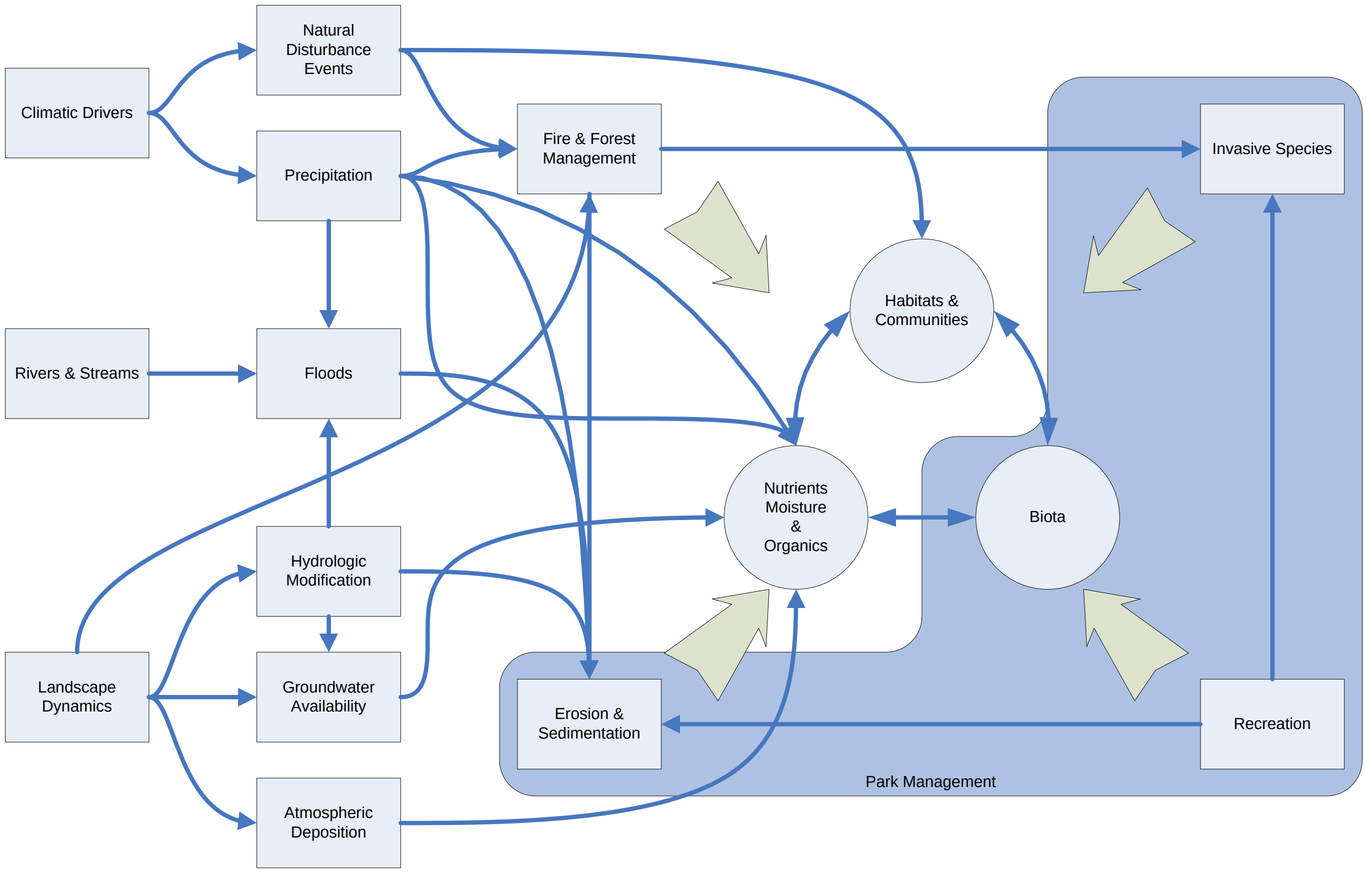

Figure A7-10. Conceptual model of ecosystem dynamics in bottomland and floodplain forests of Southeast Coast Network Parks. Rectangles indicate predominant agents of change; circles represent major components of the ecosystem (detailed in supporting text). Large greenish arrows link agents of change to the entire ecosystem, including biotic, chemical, and physical components. The dark blue area includes those agents of change or resources that are currently / actively managed by NPS.

Rivers & Streams

Isphording and Fitzpatrick (1992) described the Southeast's rivers and streams as an evolutionary laboratory. Thirty major river systems drain to the Gulf of Mexico or the Atlantic Ocean. Long isolation of these waters has produced high species richness and local endemism. Continental high points in diversity occur in fishes, salamanders, aquatic insects, crayfishes, mollusks, and freshwater snails (Isphording and Fitzpatrick, Jr. 1992, Wallace et al. 1992, Bogan et al. 1995). Taxonomic revision is ongoing in these groups, and new species are still being discovered. Systematic and genetic relatedness among the species has been used to describe biogeographic provinces and evolutionary histories (for example, Sheldon 1988). Six broad geographical provinces were based on several animal groups (fishes, mollusks, and crayfishes): the Atlantic Coastal Plain, the eastern Gulf Coastal Plain, the southern Appalachians, peninsular Florida, the Great River (Ohio-Mississippi) systems, and the trans-Mississippi region (Isphording and Fitzpatrick, Jr. 1992). The faunas of the Atlantic Coastal Plain and the eastern Gulf Coastal Plain had their origins in different parts of the southern highlands. The southern Appalachians have a high degree of endemism in isolated headwater streams. SECN parks contain systems within four physiographic provinces.

SECN stream systems have been altered by human activities, including impoundment, channelization, lowering of water tables, increased runoff, acid mine drainage, air and water pollution, sedimentation, recreation, and introduced species (including mussels, fishes, and aquatic plants) (Figure A7-11). Many examples of effects on stream biota can be cited (Hackney and Adams 1992)—nearly all major stream systems have been channelized or dammed (Adams and Hackney 1992). In the Southeast, 144 major reservoirs have been built (Soballe et al. 1992), and one-third of all Florida rivers have impoundments. The closing of the Norris Dam on the Clinch River in

Tennessee in 1936 caused a loss of 45 mussel species below the dam within 4 months (Soballe et al. 1992). The creation of the Tennessee-Tombigbee Canal is allowing mixing of formerly isolated native biota; Sheldon (1988) predicted this mixing will result in species loss through competition and interspecific hybridization. Between 1930 and 1971, 2,017 square kilometers were surface-mined in the Appalachian Highlands, leading to acidification of nearby streams and reductions in aquatic species diversity and biomass (Mulholland and Lenat 1992). Water hyacinth, a nonindigenous plant first introduced to New Orleans in 1884, had become a problem locally by 1890 and covered 80,000 hectares in Florida by 1975 (Crisman 1992). Major hydropower facilities are located upstream of three of the six river parks in the network (CONG, HOBE, CHAT).

Only 20% of the nation's freshwater communities are protected by federal laws, and of these, only 10% are east of the Mississippi (Benke 1990). Despite having the highest diversity of fish species in the United States (McAllister et al. 1986), the rivers and streams of the Southeast are little understood and only minimally protected. Lotic species (those that live in moving water), especially those of higher elevations, are most seriously affected, as their specialization to clear, fast-moving streams renders them unable to adapt to conditions caused by dredging or impoundment (Hackney and Adams 1992).

River systems in the Southeast generally follow trends as described in Vannote et al.'s (1980) River Continuum Concept. This model describes linkages between streams, floodplains, and the watersheds that they drain along a longitudinal gradient from the headwaters to the sea (Figure A7-11). The River Continuum Concept maintains that biological, physical, and chemical properties and functions of river systems and their associated floodplains follow a general pattern from their headwaters to their mouths due to changes along gradients such as elevation, geomorphology, amount of water, and the amount of light.

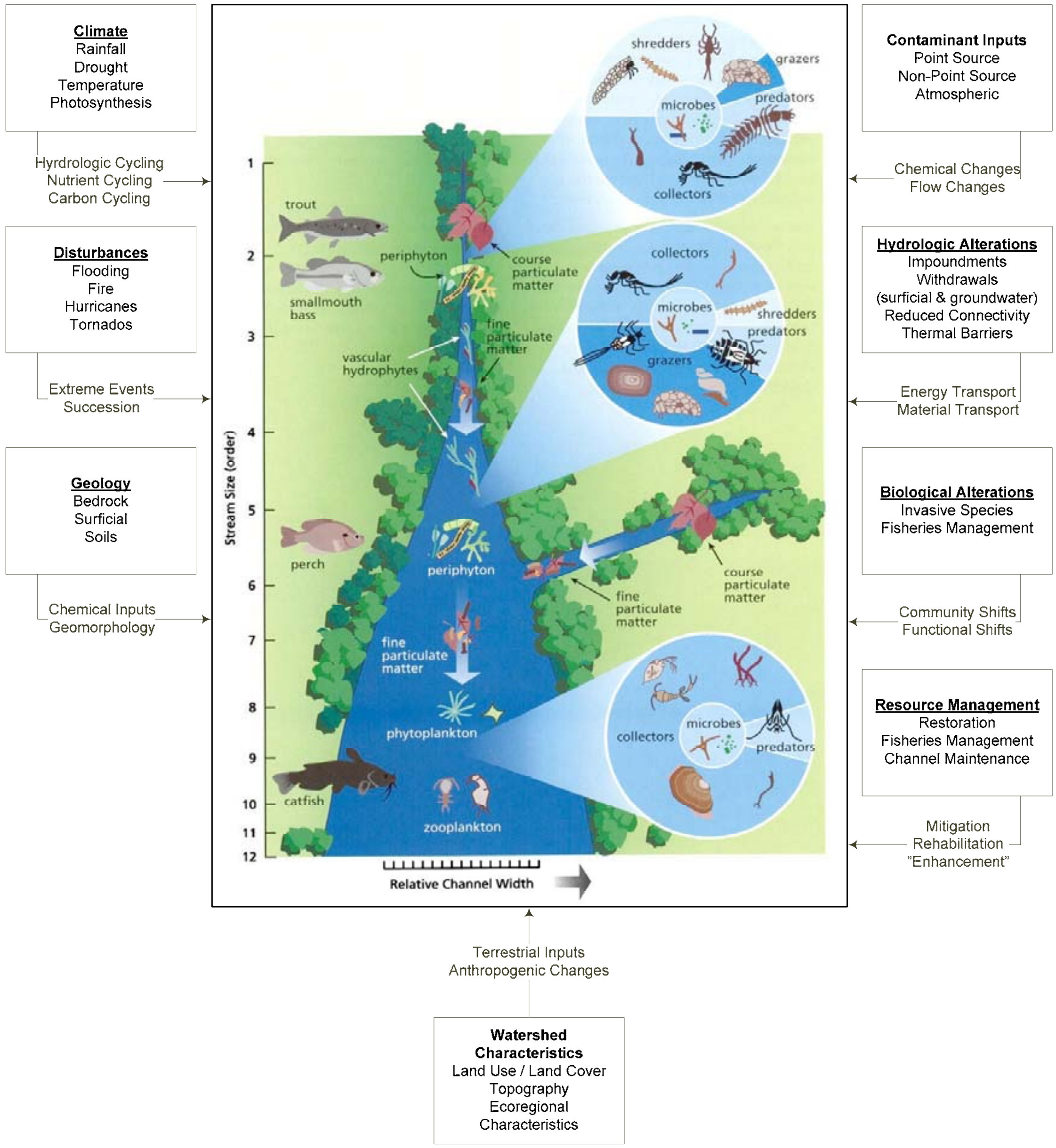

Figure A7-11. General ecosystem model for river and stream systems within the Southeast Coast Network. Modified from Vannote et al. (1980).

Southeast Coast Network parks contain significant riverine resources within three distinct zones along the river continuum—CHAT and KEMO are located in the Piedmont province, HOBE and OCMU are on the fall line, and MOCR and COSW are located within the coastal plain (Figure A7-12). Coastal parks within the network also contain smaller isolated systems.

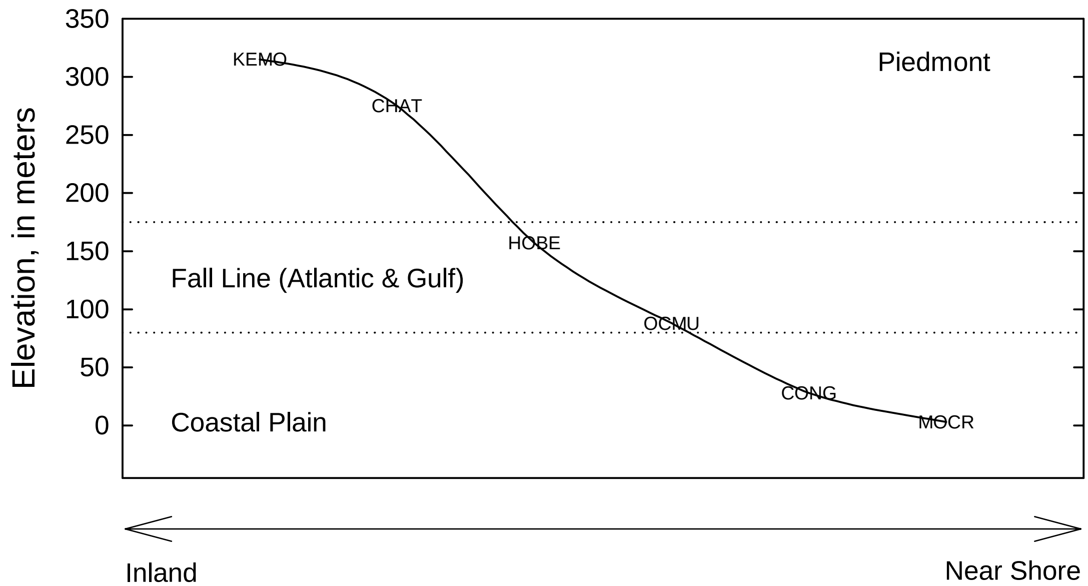

Figure A7-12. Location of the six river parks within the Southeast Coast Network along an elevation gradient. River systems within the network span two physiographic provinces (Piedmont and Coastal Plain) within the region as well as the transition zone between those provinces.

Southeast Coast Network streams and rivers are in a modified to highly modified state due to a combination of river regulation and rapid changes in land use that have resulted in extreme changes in water quality, habitat quality (through sedimentation) and aquatic community structure (Figure A7-13). Southeastern streams that were once dominated by coarse woody debris and gravel-bottom substrates have seen those substrates either cleared or buried, and many sensitive species (such as mussels) have been extirpated as a result. Although natural disturbances cause local or system-wide modifications to one or more of these components, these variations are considered to be a part of the natural state. Key processes that drive the natural system to one or more of the modified states include flow restriction and redirection, water withdrawal, species introductions, erosion (Figure A7-13).

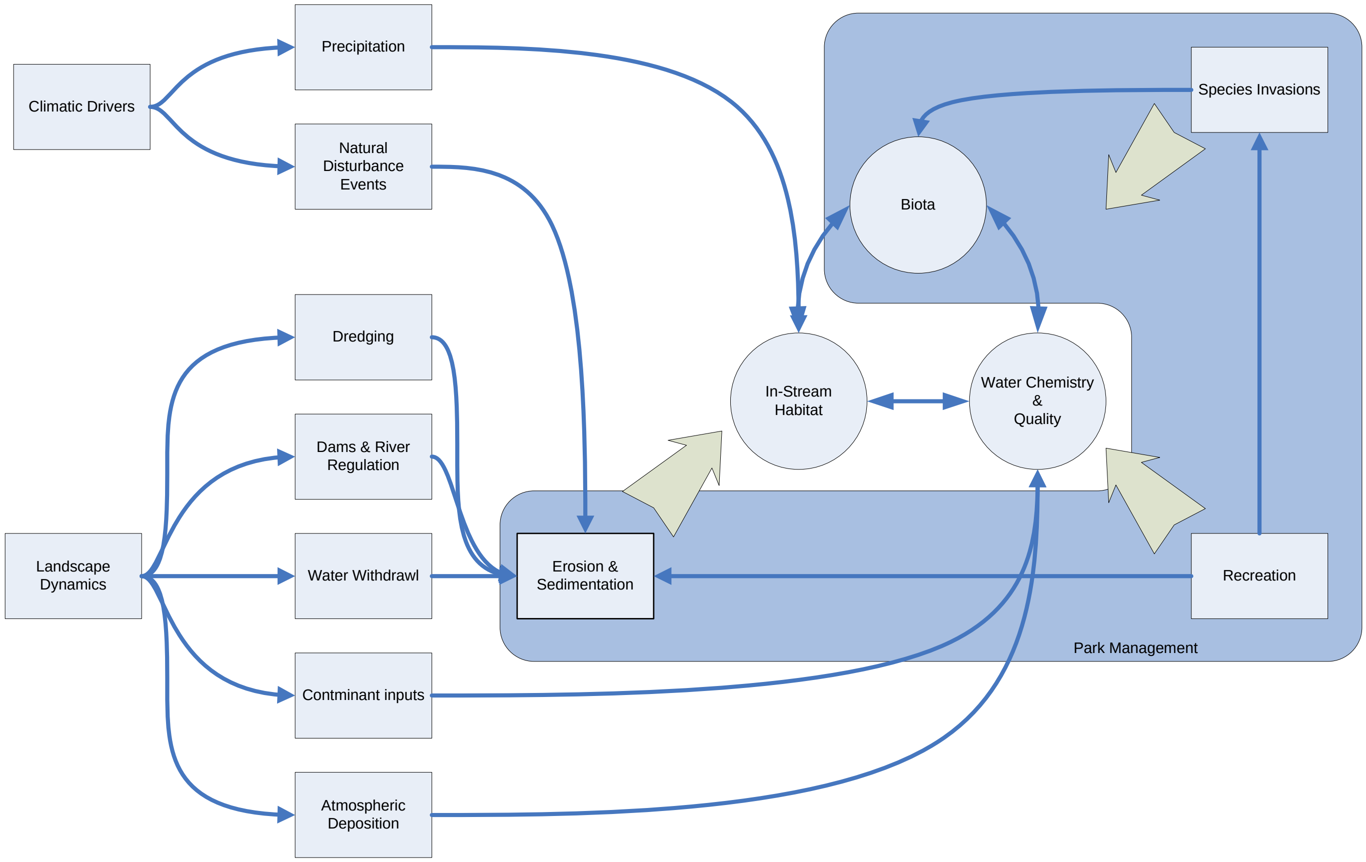

Figure A7-13. Conceptual model of ecosystem dynamics in streams and rivers of Southeast Coast Network Parks. Rectangles indicate predominant agents of change; circles represent major components of the ecosystem (detailed in supporting text). Large greenish arrows link agents of change to the entire ecosystem, including biotic, chemical, and physical components. The dark blue area includes those agents of change or resources that are currently / actively managed by NPS.

Salt Marshes & Coastal Wetlands

The Southeast region is characterized by vast expanses of coastal marshland, large beds of seagrasses, and some of the most highly productive fisheries in the country. On a global scale, a positive relationship has long been recognized between the extent of coastal wetlands and fishery landings (Turner 1977). On a smaller scale, investigations of animal distributions within estuaries have documented high densities of juvenile fishes, shrimps, and crabs in seagrass and marsh habitats compared with sites lacking bottom vegetation (Zimmerman and Minello 1984, Hoss and Thayer 1993, Peterson and Turner 1994). These patterns indicate that wetlands provide important nursery functions. Indeed, other research has shown that wetland habitats provide young fishery species with both an abundant source of food to support rapid growth and also protective cover to reduce mortality from predators (Boesch and Turner 1984, Kenworthy et al. 1988, Minello et al. 1989, Minello and Zimmerman 1991).

The linkages between wetlands and fishery productivity, however, can be complex. For example, the importance of marsh availability has only been fully recognized within the last decade. Availability of coastal marshes to fishery organisms is determined by tidal flooding patterns, the amount of marsh-water edge, and the extent of connections between interior marsh and the sea. Within the Southeast, low-elevation marshes in the northern Gulf of Mexico are flooded almost continually during some seasons and are extensively fragmented, providing maximum access for young fishery organisms. In contrast, marshes along the South Atlantic coast have relatively little marsh-water edge and appear to be infrequently flooded. The density of fishery species using the marsh surface also varies between these areas; densities in the Gulf of Mexico marshes are generally an order of magnitude greater than those on the Atlantic coast (Rozas 1993). Researchers now believe that these differences in wetland availability and use are at least partially responsible for the higher landings of estuarine-dependent species in the Gulf of Mexico compared with the South Atlantic.

One major function of wetlands is to provide food for fishery species, and there is evidence that this function also varies regionally. Historically, salt marshes were thought to contribute mainly to detrital food webs by outwelling plant debris into downstream estuaries (Nixon 1980). Such an indirect use of marsh plant production is consistent with the high elevations and large tidal regimes characteristic of Atlantic coast marshes. In the northern Gulf of Mexico, however, direct use of the marsh surface appears more common and is fostered by low marsh elevations and extensive flooding with small tidal regimes. If organisms have access to the marsh surface, primary producers such as benthic and epiphytic algae, along with abundant small consumers, provide plenty of the high-quality food necessary for young fishery species. Thus, the relative importance of different trophic pathways is probably controlled by wetland availability (McIvor and Rozas 1996).

Overlying and perhaps overshadowing these concepts of relative wetland value are the extensive rates of coastal marsh loss occurring in the Southeast, mainly in the northern Gulf of Mexico. Because of the linkages between wetlands and fishery production, we might expect dramatic declines in estuarine-dependent fisheries as marsh habitats are lost. However, over the last 20 to 30 years, productivity and landings of three dominant fishery species (brown shrimp, white shrimp, and menhaden) in the northern Gulf of Mexico have increased (Klima et al. 1990, Smith 1991). In contrast, production of these species did not increase on the Atlantic coast where wetland loss was low compared with the Gulf of Mexico. We are left with a paradox—increased production of fishery species appears correlated with the degradation of their habitat. The explanation may lie in understanding the process of wetland degradation. Wetland loss in the northern Gulf of Mexico is mainly caused by coastal submergence, canal dredging, levee construction, and erosion (Rozas and Reed 1993, Turner 1997). Concurrently, marsh flooding increases, fragmentation and habitat edge increase, zones of saline and brackish wetland expand, and connections with the sea are shortened. These processes increase the availability and value of the remaining marsh and may be supporting short-term increases in fishery production (Zimmerman et al. 1991, Rozas 1995). If this hypothesis is true, enhanced levels of fishery productivity in the Gulf of Mexico are temporary. Continued wetland loss will overcome any benefits of habitat degradation and result in future declines in fishery production dependent on these coastal wetlands.

The salt marsh ecosystem is extremely diverse, including plant, animal, and microbe communities of the marsh and plankton, invertebrates and fish in the tidal creeks, and estuaries (discussed below) (Figure A7-14). In general, plant communities follow a predictable zonation (Figure A7-15), which is largely driven by salinity and latitude. Ecosystem functions of salt marshes include the following:

- Primary productivity is generally high,
- Detrital production is both high and important for fish nurseries and linked estuarine systems,
- Detrital decomposition is the major pathway for energy flow within salt marshes
- Salt marshes can be either a source or a sink for nutrients.

Wetlands within SECN parks vary widely from intermittent interdunal pools to riparian floodplains to vast salt marshes. These systems are particularly sensitive to changes in water quantity. TIMU is the “type” location for *Spartina* salt marshes in the Southeast. Within network parks, these areas are threatened by visitor uses, commercial and recreational fishing and ecosystem drivers from inland and upstream areas (Figure A7-16). The long-term stability of salt marsh ecosystems will likely be increasingly driven by those inland and upstream processes such as water use, landscape dynamics, and their cumulative effects on incoming water quality and quantity (Figure A7-16).

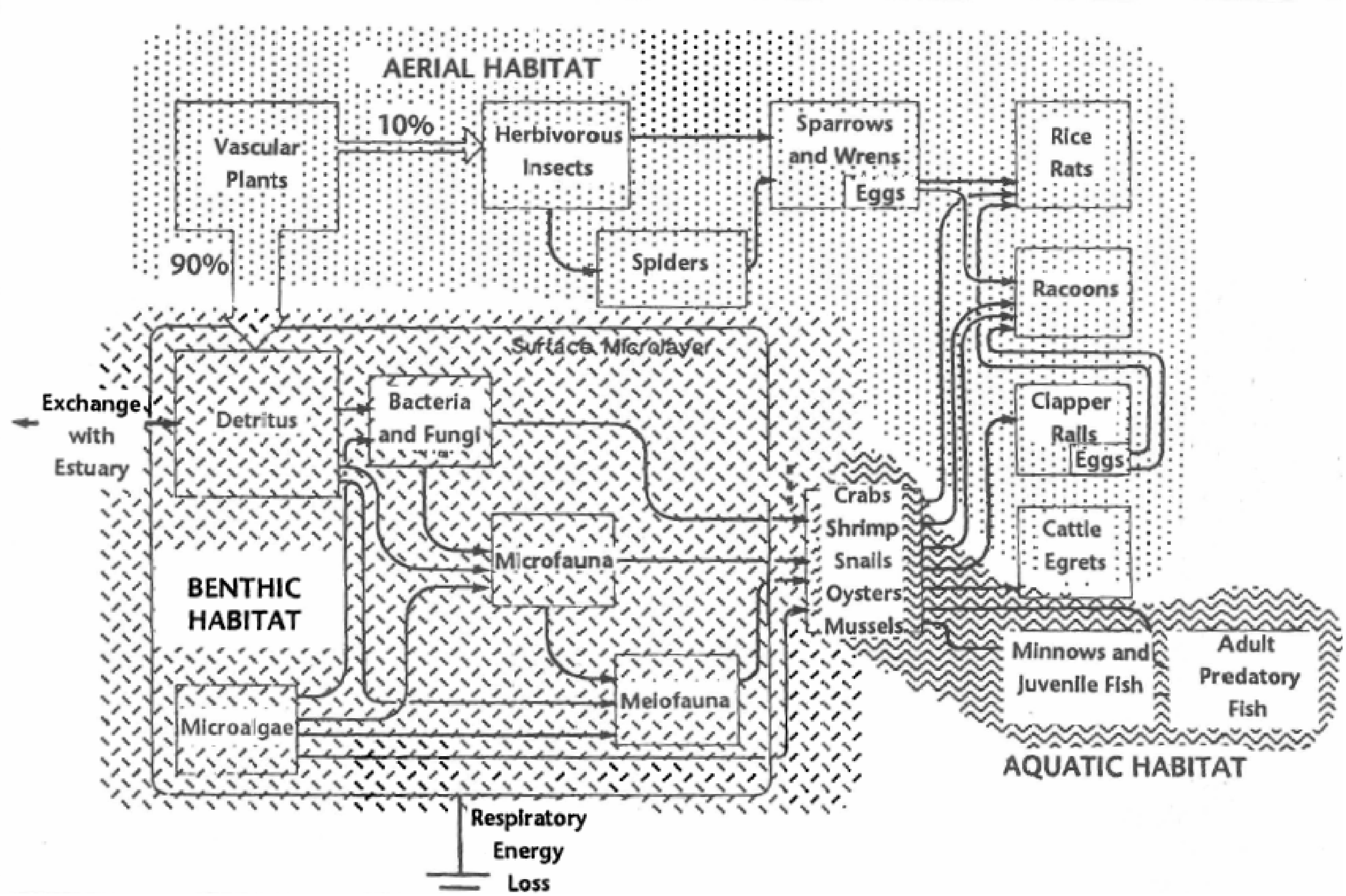

Figure A7-14. Salt marsh food web, showing the major consumer groups of the *aerial* habitat, *benthic* habitat, and *aquatic* habitat (modified from Montague and Wiegert 1990).

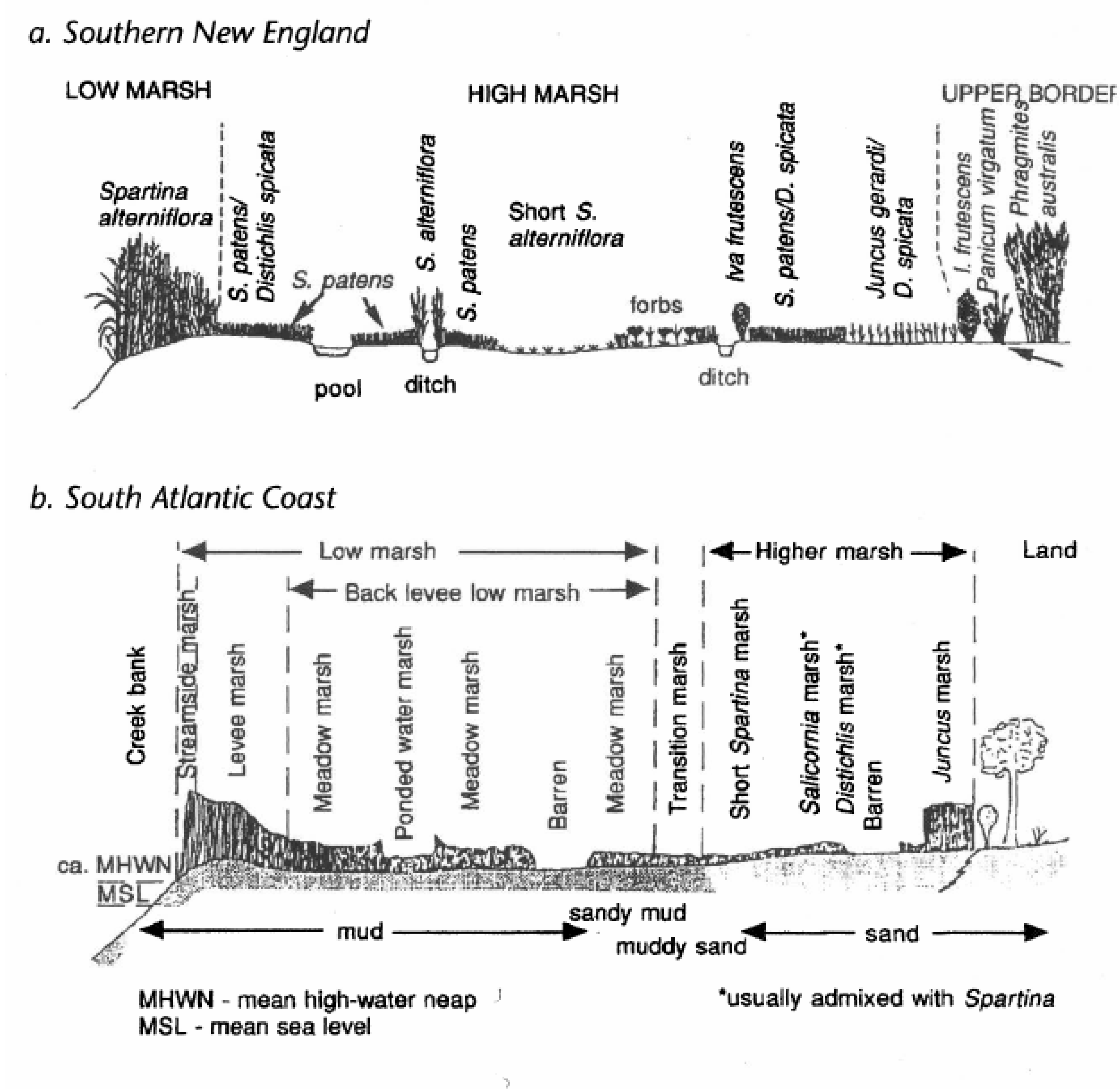

Figure A7-15. Zonation of vegetation in typical North American salt marshes. *a.* Southern New England (Niering and Warren 1980); *b.* South Atlantic Coast (Wiegert and Feeman 1990).

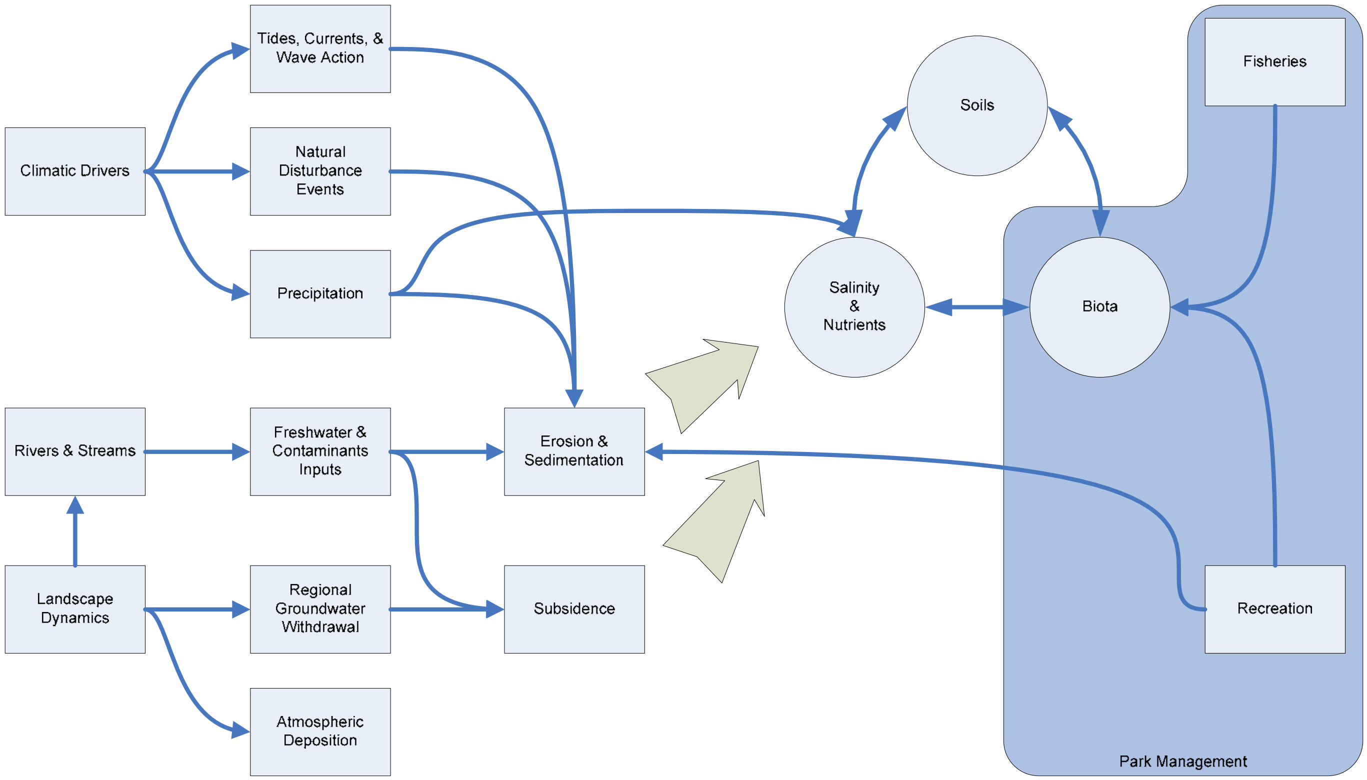

Figure A7-16. Conceptual model of ecosystem dynamics in coastal wetlands and salt marshes of Southeast Coast Network Parks. Rectangles indicate predominant agents of change; circles represent major components of the ecosystem (detailed in supporting text). Large greenish arrows link agents of change to the entire ecosystem, including biotic, chemical, and physical components. The dark blue area includes those agents of change or resources that are currently / actively managed by NPS.

Estuaries and Nearshore-Marine Systems

Most fishery species within the Southeast shelf ecosystem spend part of their life cycle in estuaries, where there appears to be an important linkage between coastal wetlands and fishery productivity. Existing data show that the overall condition of the U.S. coastal waters as fair to poor, varying from region to region and that 44% of estuarine areas in the U.S. are impaired for human use or aquatic life use. To determine the overall condition of the Nation's estuaries, EPA measured seven coastal condition indicators, including water clarity, dissolved oxygen, sediments, benthos, fish contamination, coastal wetlands loss, and eutrophication. These indicators were rated in estuaries in each region of the country (northeastern, southeastern, Gulf of Mexico, west coast, and Great Lakes regions). The condition of each resource was rated as good, fair, or poor. The indicators were combined to describe the overall coastal condition for each of the regions.

The northeastern estuaries, Gulf of Mexico and the Great Lakes are in fair to poor ecological condition, while southeastern and west coast estuaries are in fair ecological condition. Water clarity is good in west coast and northeastern estuaries, but fair in the Gulf of Mexico, southeastern estuaries, and the Great Lakes. Dissolved oxygen conditions are generally good and sediment contaminant conditions are generally poor throughout the estuaries and Great Lakes of the United States. Eutrophication in coastal waters is increasing throughout much of the United States and results in poor eutrophic conditions in the Gulf of Mexico, west coast and northeastern estuaries and in fair to good conditions in the remaining estuaries of the continental United States.

Living resources are in fair condition in estuaries throughout the United States, although small changes in water quality could cause this condition to worsen and result in a poor rating. Living resources in the Great Lakes, northeastern estuaries, Gulf of Mexico and the west coast are currently in poor condition. Contaminant concentrations in fish tissues are low throughout the estuarine waters of the United States with exceptions in selected northeastern estuaries, Gulf of Mexico estuaries and the Great Lakes. Fish consumption advisories exist throughout the Gulf of Mexico and northeastern coastal areas, although these advisories largely pertain to offshore species (e.g., king mackerel).

State assessments of water quality presented in the EPA's *National Water Quality Inventory* Report largely agree with the water quality and ecological assessment of the Nation's estuaries in the *National Coastal Condition Report*. States determine water quality conditions by comparing available water quality data to their state water quality standards. If a body of water does not fully support its designated use, such as recreation and swimming, drinking water source, or aquatic life habitat, then it is considered impaired. In 1998, states reported that 44% of estuaries and 12% of coastal shoreline in the United States (excluding Alaska) were impaired by some form of pollution or habitat degradation.

Estuarine and nearshore marine systems represent a biological continuum from riverine and salt marsh systems out to the sea (Figure A7-17) (Livingston 1990). Most parks within the network (even within the coastal parks) do not have jurisdiction within estuarine or marine systems. However, many resources within park boundaries rely on estuarine or nearshore marine systems for part of their life cycle (Figure A7-18).

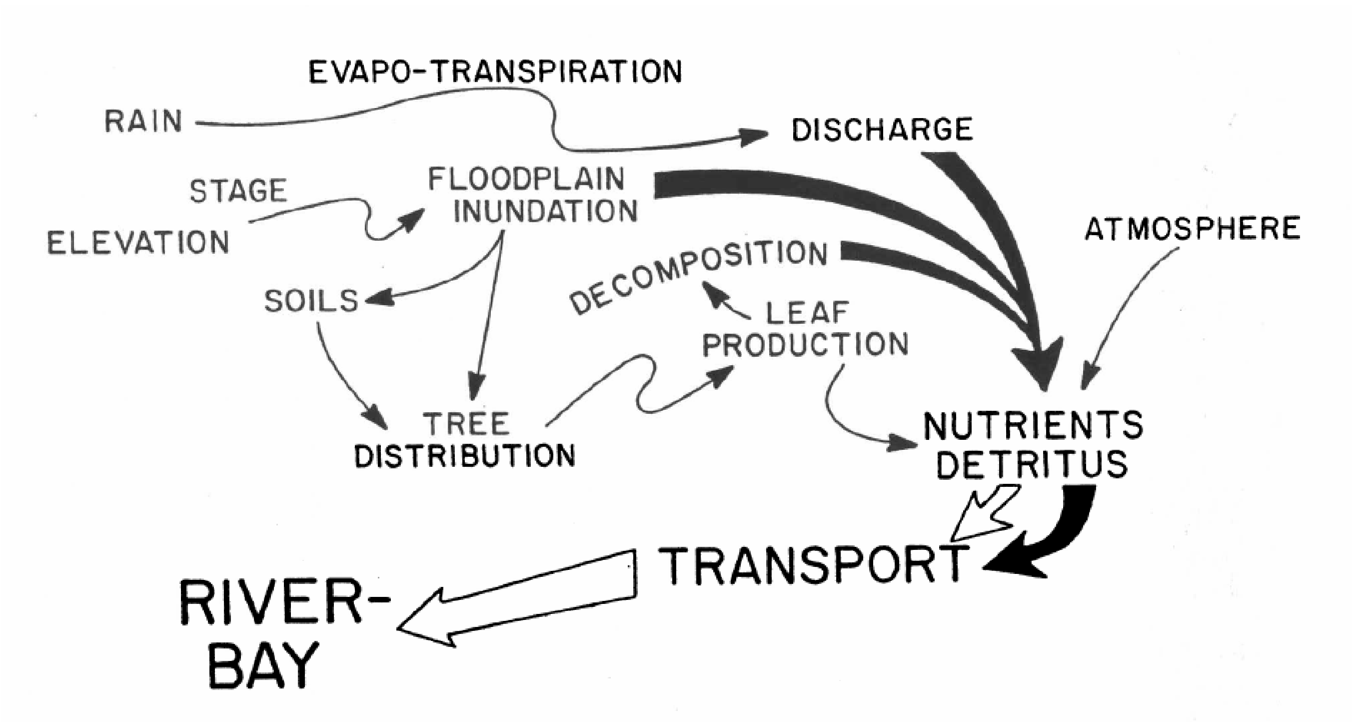

Figure A7-17. Model of the movement of nutrients and organic matter from alluvial wetlands into associated river-estuary systems (Livingston 1984).

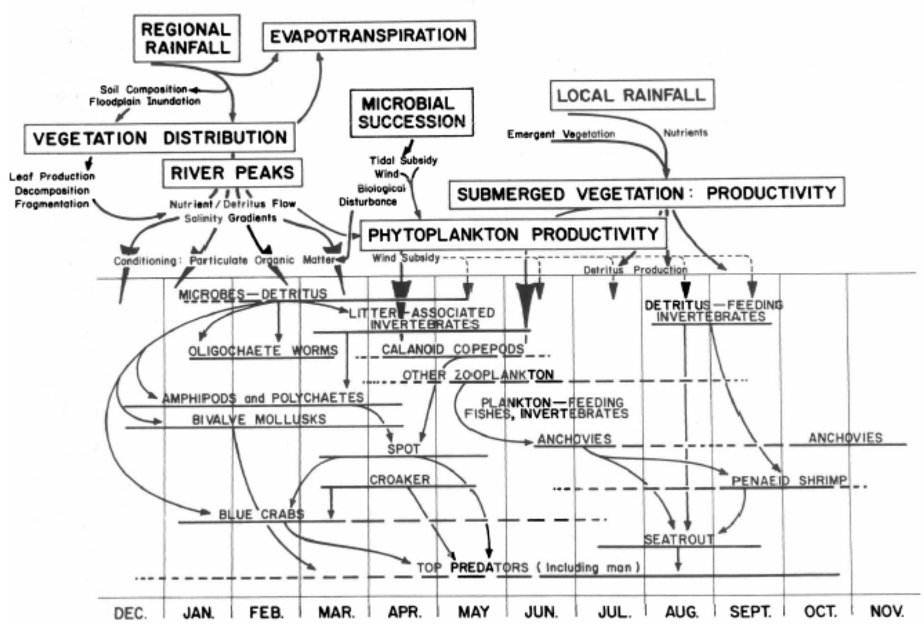

Figure A7-18. Biotic and abiotic drivers of processes in river-bay ecosystems. (After Livingston 1990).

The Southeast Coast Network contains seven parks with significant portions of estuarine and nearshore marine systems either within or adjacent to their boundaries (CAHA, CALO, FOSU, FOPU, CUIS, TIMU, and CANA). Estuarine systems are sensitive to changes in hydrology; particularly those that can affect salinity levels. The major anthropogenic agents of ecosystem change in estuarine and nearshore marine systems include coastal zone management (dredging, beach renourishment, and shoreline stabilization projects), fisheries, changing landscape dynamics, and hydrological modifications resulting from both upstream river regulation and groundwater extraction (Figure A7-19). Potential changes to the ecosystem include modified hydrology (flushing), modified disturbance regimes (flooding frequency), modified habitats (a combination of changes in sand / sediment budgets and water quality), and resultant shifts in community structures or distributions.

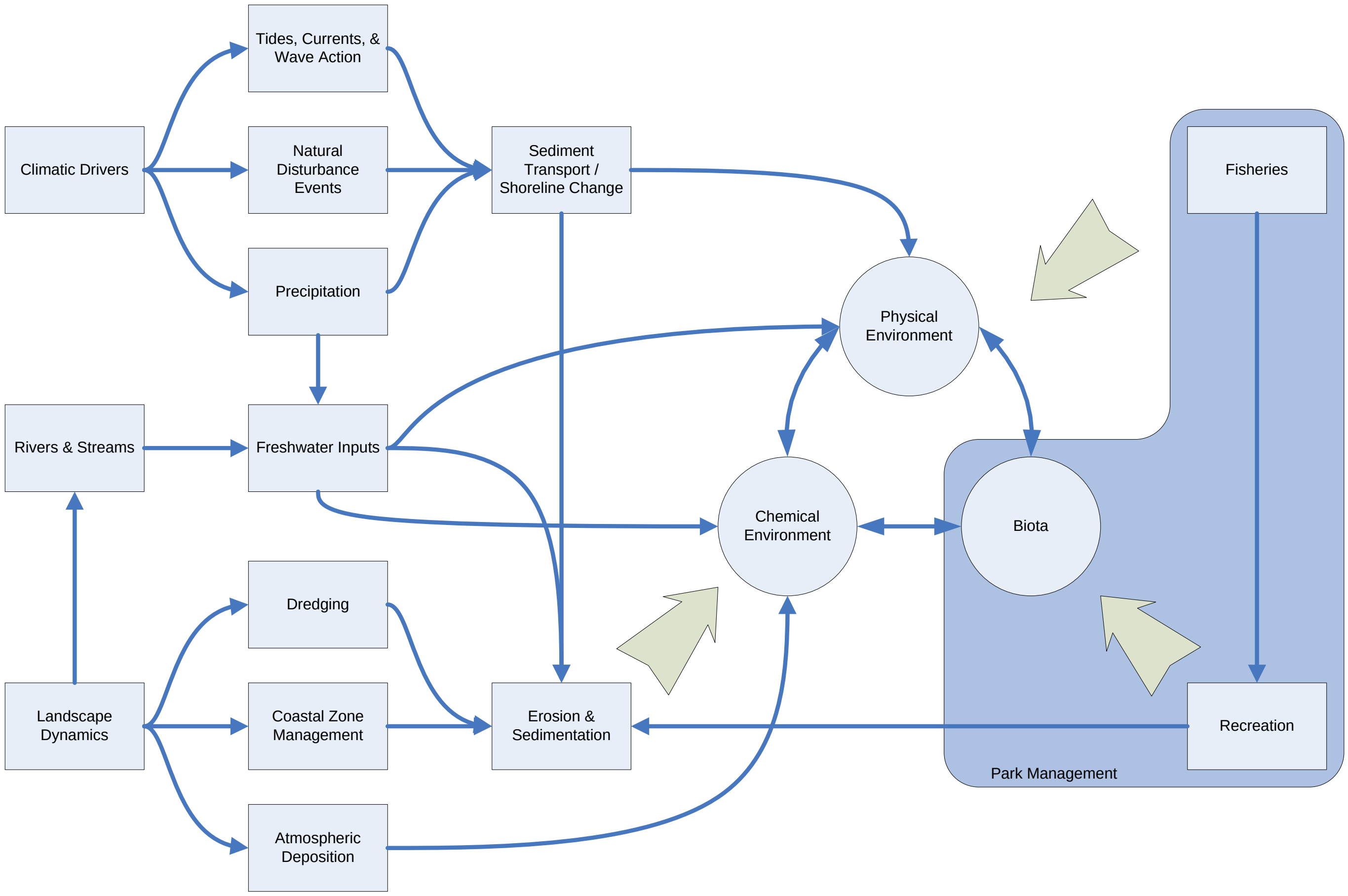

Figure A7-19. Conceptual model of ecosystem dynamics in estuaries and nearshore marine systems of Southeast Coast Network Parks. Rectangles indicate predominant agents of change; circles represent major components of the ecosystem (detailed in supporting text). Large greenish arrows link agents of change to the entire ecosystem, including biotic, chemical, and physical components. The dark blue area includes those agents of change or resources that are currently or actively managed by NPS.

Coastal Barrier Islands

The Southeast supports over 200 individual barrier islands with a total area of over 610,000 hectares (Bellis and Keough 1995). The ecosystems of these islands are diverse and dynamic, a product of regional climate, geomorphology, local sediment deposition, and the forces of ocean currents, tides, wind, salt spray, erosion, and violent ocean storms (Bellis 1992, Stalter and Odum 1993, Bellis and Keough 1995). The islands are grouped into five geographical categories: the mid-Atlantic region, extending from New Jersey to Cape Hatteras, North Carolina; the Sea Islands, bordering the coasts of South Carolina and Georgia; the Florida Atlantic; the eastern Gulf of Mexico coast; and the Louisiana-Texas Gulf of Mexico coast (Stalter and Odum 1993).

Human activities have only had a major effect on the barrier islands in the past 50 years. Eighteenth- and nineteenth-century settlements were small, scattered, and difficult to reach. Most activities were confined to forestry, livestock grazing, and subsistence agriculture, except in the Georgia and South Carolina Sea Islands, where cotton and rice plantations were widespread. The construction of bridges and causeways and the improvement of transportation in the early part of the 20th century brought new opportunities for recreation, tourism, and second-home development. Development has meant the construction of jetties and sea walls, filling and draining of marshes, and extensive dune stabilization and beach nourishment programs, all of which obstruct the natural fluctuations of the barrier island communities. Despite limited fresh water and the constant threat of storm damage, development continues at an accelerating pace (Stalter and Odum 1993). Barrier island development in the Southeast has increased more than 300% in the past 50 years, and coastal Florida's development proceeds at a rate nearly twice that of the entire Atlantic and Gulf of Mexico coasts combined (Johnson and Barbour 1990). Although there are stretches of protected barrier island beaches and dunes and intact salt- and freshwater marshes, close to half of the area of these communities is estimated to have already been lost (Noss et al. 1995).

Development, of course, has many effects. Beach traffic disturbs nesting birds and sea turtles, compacts the soil, and disrupts dune-building activities. Jetties, sea walls, inlet stabilization, and artificial dunes disrupt normal overwash activities, altering normal dune development and increasing erosion in some areas and sand deposition in others. Development within the foredune zone and forest clearing destroy natural protective barriers to salt spray and wind damage. Pollution of marshes, estuaries, and creeks is a common result of inputs of treated and untreated sewage, fertilizer runoff from developments such as golf courses, and numerous contaminants from marinas, fish-processing plants, highways, and small industries (Stalter and Odum 1993). Finally, fragmentation of vegetation interferes with natural migration patterns.

Experience with severe storm damage on coastal structures has modified development activities to some extent. Today, setback requirements in effect in many areas prohibit the destruction of the foredunes and reduce effects on beach areas. Existing structures, however, still require protection from beach migration, as well as regular, costly, beach nourishment projects (Johnson and Barbour 1990). About one-third of the barrier islands lining the Atlantic and Gulf of Mexico coasts have been protected by being set aside as parks, wildlife management areas, and national seashores (Stalter and Odum 1993). Areas that are open for development, however, are largely at risk for continued severe habitat degradation and other environmental losses. Most of the Atlantic coast of Florida is unprotected and very little natural coastline remains.

Maritime communities have decreased in areal extent since settlement, but the magnitude is known only for local areas. For example, coastal wetlands around Tampa Bay have decreased by 44% (Johnston et al. 1995). From 1950 to the present, the area of coastal wetlands along the Gulf of Mexico decreased by 20%-35% (Johnston et al. 1995); the largest losses were in Louisiana, where coastal impoundments flooded wetlands. In general, freshwater wetlands have decreased to a much greater extent than estuarine wetlands. In 1982 the Coastal Barrier Resources Act restricted the use of federal funds for development of barrier islands. An extensive monitoring system has shown that the area of undeveloped barrier islands has been stable since that law was passed (Williams and Johnston 1995). Within SECN parks, changes in plant community types have been driven by a history of ditching and draining, conversion of lands for silviculture, and extensive grazing (Figure A7-20 and Figure A7-21).

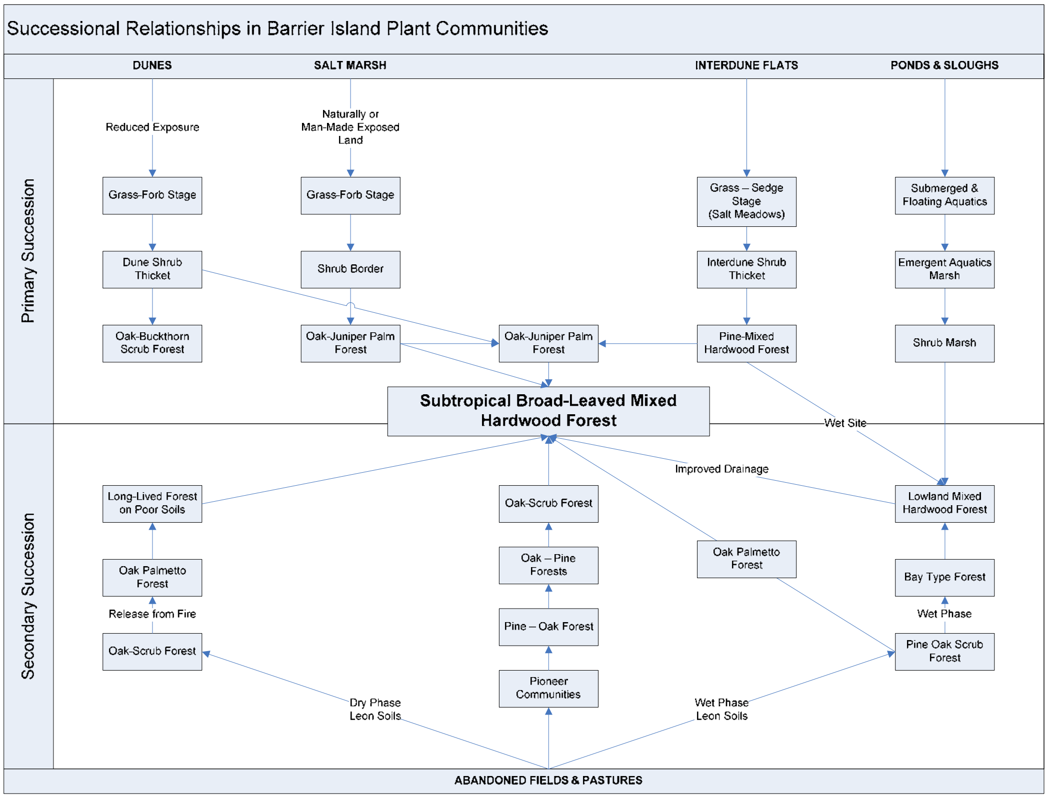

Figure A7-20. Successional relationships of plant communities on barrier islands. Modified from Hillestad et al. (1975).

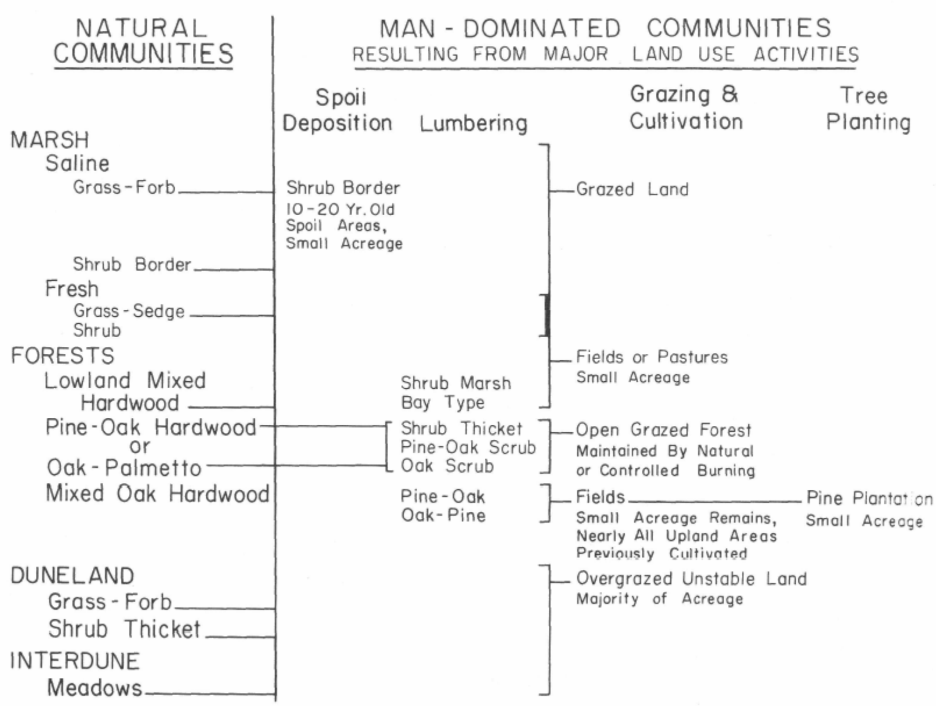

Figure A7-21. Relationship of the natural plant communities and historical land use practices on barrier islands. From Hillestad et al. (1975).

Bellis and Keough (1995) estimated that 39,000 hectares of maritime forest occurred in North Carolina, Georgia, and Florida, the three states with the best inventories. This represents an unknown fraction of the original extent of these forests. About half of the remaining forests are unprotected and likely to be developed within the next decade (Bellis and Keough 1995). The degree of human disturbance and changes within the small forest fragments that remain (for example, edge effect and the fact that fragments may not be large enough to support a population big enough to convey long-term persistence) produce declines in the numbers and species of many animals (Gaddy and Kohlsaat 1987, Bellis and Keough 1995).

Several investigators noted the inadequacy of existing data for detection of trends. Bellis and Keough (1995) suggested the need for a complete survey and assessment of maritime forests. Besides effects of development and nonindigenous species, maritime communities will probably be influenced by sea-level rise and drawdown of freshwater supplies (Bellis and Keough 1995). Daniels et al. (1993) modeled the influence of sea-level rise on endangered species in South Carolina and showed that 52% of the regionally endangered species were found within 3 meters of current mean sea level and that several scenarios of sea-level rise would drastically reduce the habitat for these species.

Coastal dunes and barrier island ecosystems are major features at CAHA, CALO, CUIS, and CANA. Future land acquisitions at TIMU might result in the addition of dune habitats there as well. Coastal dunes are particularly important due to the fact that (a) they support a wide variety of sensitive or protected species, (b) they are fragile, (c) they are particularly threatened by visitor uses, and (d) they play a significant role in the overall stability of the island.

Large numbers of migratory and nesting bird species are found on barrier islands (Stalter and Odum 1993); for example, 350 species have been recorded on barrier islands in North Carolina alone (Parnell et al. 1992). Coastal marshes are critical to overwintering populations of many waterbirds. In addition, migration routes of many raptor species include southeastern barrier islands. Neotropical migrants use the islands as a point of departure and arrival in their travels to and from their winter habitats in the tropics (Stalter and Odum 1993).

Many birds have been negatively affected by development and human encroachment. Species that nest in bare sand can be disturbed by pedestrian and off-road vehicle traffic and by the construction of artificial dunes. Harrington (1995) reported that for 27 species of eastern shorebirds, 12 had stable populations, 1 was increasing, and 14 were decreasing. Surveys initiated off the North Carolina coast in the early 1970's tracked the fluctuations in nesting bird populations (Parnell et al. 1992). Eight species were increasing strongly (brown pelican, cattle egret, white ibis, glossy ibis, laughing gull, herring gull, royal tern, and Sandwich tern), three were increasing (yellow-crowned night-heron, great black-backed gull, and caspian tern), four were declining (gull-billed tern, common tern, least tern, and black skimmer), and seven were presumed stable. Some of the species have even shifted locations; Parnell et al. (1992) suggested that cutting of coastal swamps during the last 50 years resulted in movement to the estuaries. Further, creation of new habitat from dredged material may have caused populations to shift from one estuary to another.

Stalter and Odum (1993) listed nine endangered species of birds that are wholly or partially dependent on habitat on southeastern barrier islands: whooping crane, Eskimo curlew, bald eagle, Arctic peregrine falcon, eastern brown pelican, Cape Sable seaside sparrow, Bachman's warbler, Kirtland's warbler, and red-cockaded woodpecker. These species use the barrier islands in a variety of ways: nesting (five species), migration (four species), wintering (five species), feeding (seven species), and resting-roosting (seven species). Stalter and Odum (1993) attributed population losses in these species to development (direct loss of nesting, resting, and foraging habitat), dredging and filling of marshlands (loss of community structure and composition used by the birds), pollution, and direct disturbance on recreational beaches.

Five species of sea turtles are found in the open ocean and coastal waters of the Southeast, and all nest on open beaches: the green sea turtle (status: endangered/threatened; U.S. Department of Commerce 1994), the hawksbill (endangered), Kemp's ridley (endangered), the leatherback (endangered), and the loggerhead (threatened). Sea turtles are difficult to census in open waters and, because of the concentration of female turtles nesting on the beach strand and the apparent faithfulness of their return to specific beaches, the number of nesting females is considered the single best indicator of population trends (Committee on Sea Turtle Conservation 1990). The Kemp's ridley nests annually, but the other species nest less regularly. Long-term data sets (that is, over a decade of observations) are essential to detecting trends (Committee on Sea Turtle Conservation 1990). The dependence of sea turtle species on the narrow beach strand also makes them vulnerable to a host of human-caused problems, including beach development and recreation, artificial lighting (which disorients hatchlings), and increases in nest predators such as raccoons. Recently, federal law has mandated that shrimp trawlers use turtle exclusion devices, which should decrease mortality in a critical life stage for reproduction (Committee on Sea Turtle Conservation 1990).

Population estimates are available for only two of the five species of sea turtles (U.S. Department of Commerce 1994): 20,000-28,000 loggerheads and 400-500 green sea turtles nest in the United States. Although the number of nesting loggerheads has declined by 3% annually at a site in Georgia and by 26% during the 1980's at a site in South Carolina, it has increased at several sites in Florida (Committee on Sea Turtle Conservation 1990, Dodd, Jr. 1995a). Summed across the Southeast, loggerheads increased from 1982 to 1990 and decreased from 1990 to 1993 (Dodd, Jr. 1995a), although the recent decline has been relatively mild, leaving the species at higher levels than in the early 1980's. A recent review concluded that the overall status of loggerhead population size was stable (U.S. Department of Commerce 1994). This study also concluded that there was inadequate data to report an overall trend in green sea turtle populations, but numbers at one Florida site had increased from 1971 to 1989, and the species is presumed to be recovering. The green sea turtle was drastically reduced by fishing (it was served in turtle soup) during the early 1900's.

At one study site in Mexico, Kemp's ridley is presumed to have declined sharply from 1947 to 1990, to 1% of original levels (Committee on Sea Turtle Conservation 1990). Data collected at that site from 1977 to 1990 suggested a continued but much less drastic downward trend. Very few hawksbills and leatherbacks nest in the United States, and data are inadequate for precise statements of trends of these species, although expert opinion holds that the hawksbill is declining (U.S. Department of Commerce 1994).

Within Network parks, the primary drivers of ecosystem change include invasive species (plant and animal), visitor uses, shoreline erosion. Many of these drivers are influenced by one or more factors outside of park boundaries linked to on-shore or upland landscape dynamics, regional climatic drivers and current and historical fire / forest management practices.

Overgrazing by exotic animals is also a problem on barrier islands, not only because of a large white-tailed deer population but also because of the large numbers of feral animals introduced to the islands, including horses, cattle, goats, pigs, and sheep (Stalter and Odum 1993). Eradication of some of the larger feral species has been successful on some islands, but other introduced animals, especially feral dogs and cats, negatively affect small mammal populations. Other introduced species include European rats and nutria (Stalter and Odum 1993). Two parks within the network (CUIS and CALO) have managed feral horse populations that are significant drivers physical, biological, and chemical components of barrier island processes.

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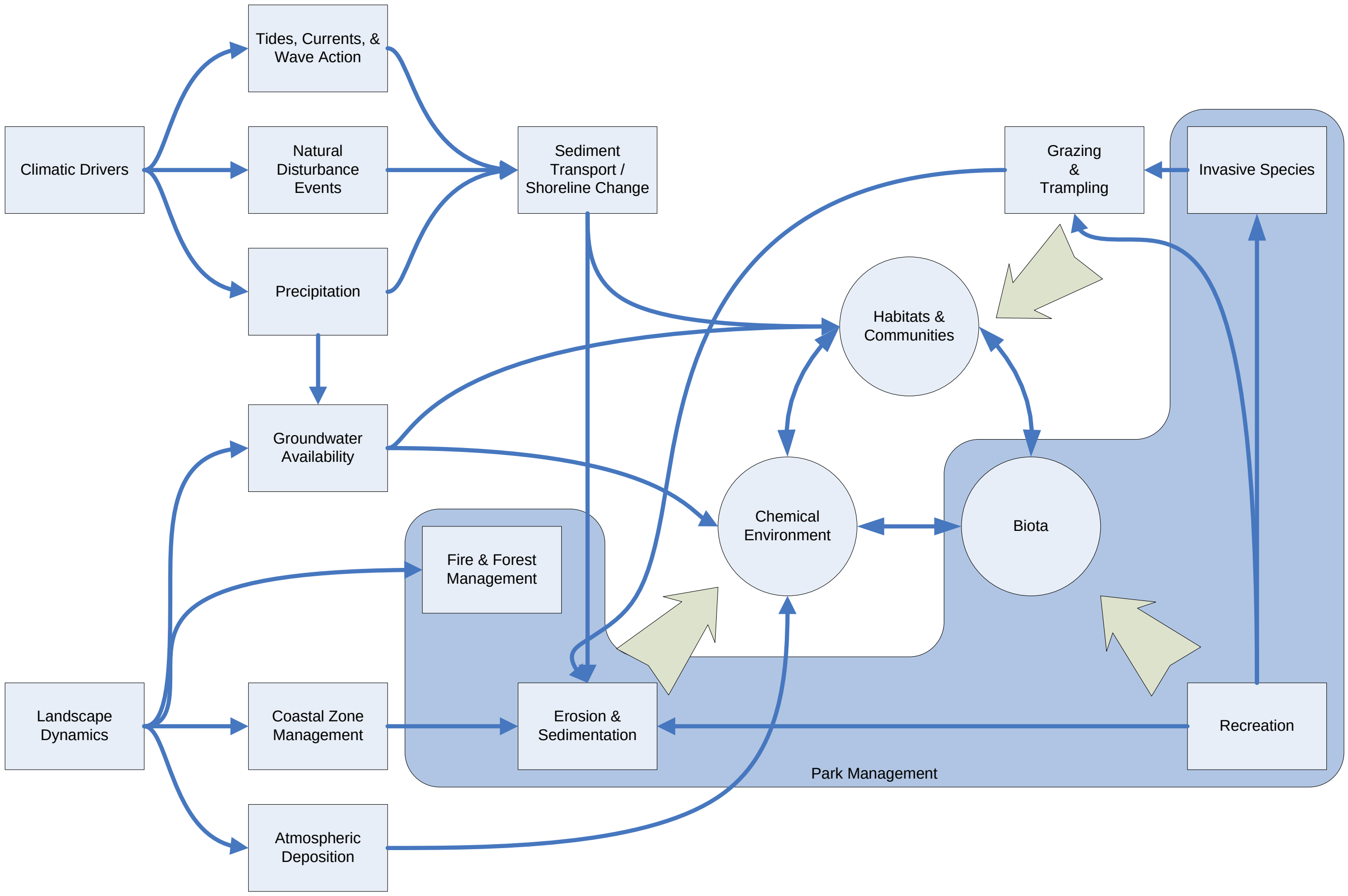

Figure A7-22. Conceptual model of ecosystem dynamics in coastal barrier islands of Southeast Coast Network Parks. Rectangles indicate predominant agents of change; circles represent major components of the ecosystem (detailed in supporting text). Large greenish arrows link agents of change to the entire ecosystem, including biotic, chemical, and physical components. The dark blue area includes those agents of change or resources that are currently / actively managed by NPS.

Ecosystem Components and Key Agents of Change

Ecosystem Components

Ecosystem components are divided into three broad categories: Environmental Setting, Park Resources, and Agents of Change. Components in the "Environmental Setting" category include those that provide the primary drivers of ecosystem structure, function, and composition. In most cases they are not actively managed by the parks due to the spatial and time scales involved (i.e., water, air, geologic, and weather resources). Park resources refer to those that are managed at one or more spatial and temporal scales ranging from individuals to ecosystems. Agents of change include both natural and anthropogenic "drivers" of ecosystem change.

Environmental Setting

Floridan Aquifer System

The Floridan aquifer system is one of the most productive aquifers in the world. It consists of a thick sequence of carbonate rocks (limestones and dolomites) that underlie all of Florida, southern Georgia, and small parts of adjoining South Carolina and Alabama; a total area of about 100,000 mi2. An estimated 4.0 billion gallons per day of water was withdrawn from the aquifer system in 2000, and, in many areas, it is the sole source of freshwater (Johnston and Bush 1988).

In addition to water supply, the Floridan is being used increasingly for aquifer storage and recovery systems, in which freshwater is injected into more saline zones of the aquifer and stored for later use. Moreover, in several places where the aquifer system contains saltwater, such as along the southeastern coast of Florida, treated sewage water and industrial wastes are injected into it.

The aquifer system generally thickens seaward from a thin edge near its northern limit to a maximum of about 3,500 ft in southwestern Florida. In most places, the system consists of the Upper and Lower Floridan aquifers separated by a less permeable confining unit (the "middle confining unit") that restricts movement of water between the two aquifers. Much of the aquifer system is overlain by an upper confining unit that, where present, limits the amount of recharge to the system. Where the upper confining unit is thin or absent, recharge is plentiful and ground-water circulation is high.

In these areas of high recharge and vigorous circulation, ground water readily dissolves the carbonate rocks that make up the aquifer system, creating large and highly permeable conduits that store and transmit tremendous volumes of ground water. These large conduits are the cause for the many first-magnitude springs—those with a flow of 100 cubic feet per second or more—that issue from the aquifer system.

Ground-water withdrawals have resulted in long-term regional water-level declines of more than 10 ft in three broad areas of the flow system: (1) coastal Georgia and adjacent South Carolina and northeast Florida; (2) west-central Florida; and (3) the Florida panhandle. In these and a number of other coastal areas, ground-water withdrawals have reversed the generally seaward direction of ground-water flow, creating the potential for saltwater intrusion from the Gulf of Mexico or Atlantic Ocean or from deep parts of the aquifer that contain saltwater.

The transition between freshwater and saltwater in the Floridan aquifer system is illustrated by the distribution of chloride in water in the Upper and Lower Floridan aquifers. Although large areas of the Upper Floridan aquifer contain water with a chloride concentration less than 250 mg/L, much of the Lower Floridan aquifer contains water with chloride concentrations that exceed the 250 mg/L drinking-water limit, which has limited the aquifer's use for water supply. In general, chloride concentrations in the Upper Floridan aquifer are related to ground-water-flow conditions and proximity to the coast. In areas where the upper confining unit is thin or absent, fresh ground-water circulation rates are high and chloride concentrations tend to be low (less than 250 mg/L). Where the flow system is tightly confined, flow is more sluggish and chloride concentrations in the aquifer are higher. This is the case in Florida south of Lake Okeechobee, where the Upper Floridan aquifer is extensively confined and ground-water flow is quite sluggish. Because of the slow movement of ground water in the area, it is thought that residual seawater that entered the aquifer during the Pleistocene when sea level was higher than its current level has not been completely flushed out by modern freshwater (Johnston and Bush 1988, Sprinkle 1989, Reese 1994, Reese 2000, Reese and Memberg 2000).

The anomalously high concentrations of chloride along the St. Johns River and the eastern coast of Florida are thought to be the result, in varying amounts, of two processes (Sprinkle 1989): (1) incomplete flushing by the modern-day freshwater flow system of residual seawater that invaded the aquifer during high sea-level stands of the Pleistocene; and (2) upward flow of brackish water from the underlying Lower Floridan aquifer along fracture zones in the aquifer system. The high chloride concentrations in the western panhandle of Florida also may have resulted from incomplete flushing of residual seawater from Pleistocene highstands. The generally low chloride concentrations in the Upper Floridan aquifer along the Georgia coast have been attributed to the thick confining unit that overlies the Upper Floridan aquifer in that area.

The confining unit has created relatively high groundwater heads that have kept the freshwater-saltwater interface offshore. In fact, freshwater flow has been observed to extend as far as 50 mi offshore of southeast Georgia (Johnston et al. 1982). However, along the coast in South Carolina and extending to Tybee Island, Georgia, high chloride concentrations in water from the aquifer system are attributed to intrusion of offshore saltwater caused by large ground-water withdrawals from the Upper Floridan aquifer in the Savannah, Georgia, and the Hilton Head

Island, South Carolina, areas. Saltwater most likely enters the aquifer system by lateral intrusion from offshore areas combined with some downward vertical leakage of seawater to the Upper Floridan aquifer where the overlying confining unit is thin or absent (Krause and Clarke 2001).

Ground-water temperature and geochemical data from the Lower Floridan aquifer in the south Florida area suggest that a geothermally driven circulation of ground water occurs in the Lower Floridan aquifer in that area (Kohout 1965, Meyer 1989, Sanford et al. 1998). In this circulation, cold, dense ocean water enters the Lower Floridan aquifer where it is in direct contact with the ocean. As the seawater moves landward, it is warmed by geothermal heat generated naturally below the base of the thick Floridan aquifer system that underlies the Florida peninsula. The geothermal heating lowers the density of the seawater, causing it to rise where it is diluted and transported back to the ocean with seaward-flowing water of lower salinity.

Northern Atlantic Coastal Plain Aquifer System

The Northern Atlantic Coastal Plain encompasses a land area of about 50,000 mi^2 extending from Long Island, New York, southward to the North Carolina-South Carolina border. The Coastal Plain is underlain by a seaward-thickening wedge of predominantly unconsolidated sediments that increases in thickness from the Fall Line, which is the inland limit of the Coastal Plain, eastward toward the Atlantic Ocean. The Fall Line is so named because of the prevalence of falls and rapids in streams that cross the contact between the hard rocks of the Piedmont Plateau to the west and the less-resistive sediments of the Coastal Plain. The sediment wedge reaches a maximum onshore thickness of about 10,000 ft at Cape Hatteras, North Carolina, but exceeds 7.5 miles in thickness offshore from New Jersey and the Delmarva Peninsula (Trapp, Jr. and Meisler 1992). The sediments are mostly gravel, sand, silt, and clay, and have been subdivided into an aquifer system that consists of a vertical sequence of highly permeable aquifers separated by less permeable confining units. Ground-water withdrawals from the aquifer system total more than a billion gallons per day, making it one of the most productive aquifer systems in the United States.

Saltwater underlies freshwater in eastern parts of the regional aquifer system. The transition zone between freshwater and saltwater was delineated throughout the aquifer system in the early 1980s by using geochemical and geophysical data collected at more than 500 locations (Meisler 1989). The transition zone was defined as the zone of water with chloride concentrations from 250 mg/L to 18,000 mg/L. Generally, chloride concentrations increase in the seaward direction of each aquifer and with depth from the shallowest to the deepest aquifers. Waters within the transition zone probably were produced by the mixing of fresh ground water with either seawater or highly concentrated brines. In the area from Virginia to New Jersey, some of the water samples showed the presence of chloride at concentrations greater than those in seawater, suggesting that the transition zone in that area is largely a mixture of freshwater with brine. For example, the chloride-concentration profile at well Virginia 57 shows a maximum chloride concentration of nearly 27,000 mg/L, which is about 8,000 mg/L greater than that of seawater. The most likely source of the brines appears to be the leaching of ancient evaporite deposits of probable early Jurassic age beneath the Continental Shelf and Slope (Meisler 1989, Knobel et al. 1998).

Two striking features of the transition zone within the regional aquifer system are its large vertical thickness and substantial horizontal width. The thickness of the transition zone ranged from 400 to 2,200 ft, whereas the width of the transition zone was as much as 40 miles in some areas. The development of the broad transition zone has been attributed to the cyclic movement of saltwater caused by global sealevel fluctuations that resulted in repeated advance and retreat of the freshwater-saltwater interface during at least the last 900,000 years (Meisler et al. 1985). As the sea level rose, saltwater invaded the aquifer sediments and mixed with freshwater. As the sea level declined, the fresher water advanced seaward, and the process of mixing continued. Repeated advance and retreat of the saltwater produced a broad zone of mixed waters in which saltwater predominates in the deeper and seaward parts, and freshwater predominates in the shallower and landward parts (Meisler et al. 1985).

The depth to the top of the transition zone is shallowest in North Carolina and deepens northward, reaching its greatest depths—as much as 2,800 ft below sea level—in Maryland and along the coast of New Jersey. Moreover, ground water containing chloride concentrations of less than 5,000 mg/L has been found as much as 55 mi from the New Jersey coast, but extends progressively shorter distances from the coast southward to Virginia and North Carolina (Meisler 1989). The occurrence of the transition zone at great depths in New Jersey and Maryland and the occurrence offshore of water considerably fresher than seawater have been attributed to long periods when sea levels generally were lower than at present (Meisler 1989). Overall, the average sea level during the past 900,000 years is estimated to have been about 150 ft lower than the present sea level. It has been hypothesized that, at least in some areas, the transition zone may not be in equilibrium with the present-day sea level, but may still be moving landward and upward to adjust to the present sea level (Meisler et al. 1985, Pope and Gordon 1999).

Air Resources – modified from United States Environmental Protection Agency (2003)

Ozone is not emitted directly into the air but formed by the reaction of volatile organic compounds (VOCs), nitrogen oxides (NO_x), and other chemical compounds in the presence of heat and sunlight, particularly in hot summer weather. Chemicals such as those that contribute to formation of ozone are collectively known as ozone "precursors." Particulate matter is emitted directly, and is also formed when emissions of NO_x, SO_2, and other gases react in the atmosphere.

With decreases in emissions of VOCs and other ozone precursors, 8-hour ozone concentrations fell by 11 percent nationally between 1982 and 2001 (U.S. Environmental Protection Agency 2002c). All regions experienced improvement in 8-hour ozone levels during the last 20 years except the North Central region, which showed little change. However, in 2001 more than 110 million people lived in counties with concentrations higher at times than the 8-hour standard for ozone (U.S. Environmental Protection Agency 2002c). Southern California, the eastern U.S., and many major metropolitan areas have continuing ozone problems.

In 2001, some 73 million people lived in counties where monitored air quality at times exceeded the standard for fine particulate matter ($PM_{2.5}$)—those particles less than or equal to 2.5 micrometers (µm) (U.S. Environmental Protection Agency 2002c). Concentrations of $PM_{2.5}$ vary regionally. California and much of the eastern U.S. have annual average $PM_{2.5}$ concentrations higher than the level of the annual $PM_{2.5}$ standard. The number of people living in counties with air quality levels that exceed the standards for ozone and PM signals continuing problems.

Pollution is impairing visibility in some of the nation's parks and other protected areas. In 1999, average visibility for the worst days in the East was approximately 15 miles. In the West, average visibility for the worst days was approximately 50 miles in 1999 (U.S. Environmental Protection Agency 2002c). Particulate matter is the major contributor to reduced visibility, which can obscure natural vistas. Without the effects of pollution, the natural visibility in the U.S. is approximately 47 to 93 miles in the East and 124 to 186 miles in the West. The higher relative humidity levels in the East result in lower natural visibility.

Two of the key pollutants that contribute to the formation of particulate matter—SO_2 and NO_x—react in the atmosphere with water, oxygen, and oxidants to form acid droplets. Rain, snow, fog, and other forms of precipitation containing the mixture of sulfuric and nitric acids fall to the earth as acid rain (wet deposition). The particles also may be deposited without precipitation, known as "dry deposition." Wet sulfate deposition has decreased substantially—20 to 30 percent—throughout the Midwest and Northeast, where acid rain has had its greatest impact, between the periods 1989-1991 and 1999-2001. During the same period, wet nitrogen deposition decreased slightly in some areas of the eastern U.S. but increased in other areas, including those with significant agricultural activity (U.S. Environmental Protection Agency 2002a).

In addition to the six criteria pollutants, the Clean Air Act identifies 188 toxic air pollutants to be regulated. Among those pollutants are benzene, found in gasoline; perchloroethylene, emitted from some dry cleaning facilities; and methylene chloride, used as a solvent by a number of industries. Often referred to as "air toxics," these are pollutants that may cause cancer or other serious health effects—reproductive effects or birth defects, for example—and may also cause adverse ecological effects.

Because there is currently no national monitoring network for toxics, concentrations of toxic air pollutants cannot be quantified on a comprehensive, national level. Data from several metropolitan areas do show downward trends in selected toxic air pollutants. For example, the levels of benzene measured at 95 urban monitoring sites decreased 47 percent from 1994 to 2000 (U.S. Environmental Protection Agency 2002c). Although data and tools for assessing the impacts of air toxics are limited, available evidence suggests that emissions of air toxics may still pose health and ecological risks in certain areas of the U.S. (U.S. Environmental Protection Agency 2002d).

Many health effects are associated with breathing polluted air, but air also transports pollutants and deposits them onto soils or surface waters, where they can potentially affect plants, crops, property, and animals. Toxic substances in plants and animals can move through the food chain and pose potential risks to human health. Airborne mercury from incineration, for example, can settle in water and contaminate fish. People and other animals higher on the food chain (e.g., bald eagles, bears, and cougars) that eat contaminated fish are then exposed to potentially harmful levels of mercury, which is known to affect the nervous system.

Direct exposure to ozone under certain conditions can be harmful to plants and forests; it reduces overall plant health and interferes with the ability of plants to produce and store food. Such weakened plants are in turn more susceptible to harsh weather, disease, and pests. Through its effects on plants, ozone can also pose risks to ecological functions such as water movement, cycling of mineral nutrients, and habitats for various animal and plant species. Airborne particles also can have an adverse impact on vegetation and ecosystems (U.S. Environmental Protection Agency 2002c).

Increased acid levels damage soils, lakes, and streams, rendering some waterbodies unfit for certain fish and wildlife species. Indirect effects of acid deposition are also responsible for damage to forest ecosystems. Excess deposition of acid ions in the soil causes calcium and other essential plant nutrients to be leached from the soil, and thus no longer available to sustain normal plant growth and maintenance. The calcium depletion also causes a scarcity of worms and other prey, affecting the ability of some birds to lay eggs and bring them to term.

Acid ions also can increase the movement of aluminum in soil, which competes with calcium and other nutrients in plant roots during absorption, further limiting plant growth. Acid deposition can also produce elevated levels of aluminum in waterbodies. This results either from direct deposits acidifying the waterbody itself or from water passing through soil that is high in aluminum and then entering the waterbody from adjacent terrestrial systems. Those elevated levels of aluminum in water can be toxic to fish and other aquatic life (U.S. Environmental Protection Agency 2002a).

The nitrogen in acid rain is one of the sources contributing to the total amount of nitrogen in terrestrial and aquatic systems. Although nitrogen is a necessary nutrient in productive ecosystems, too much nitrogen in terrestrial systems can cause changes in biodiversity. In aquatic systems, it fuels excessive growth of algae in coastal waters. When the dense algal blooms die, bacteria decay them. That process uses up the oxygen that is needed by fish to survive.

Geologic Resources

Soils

Island Soils – modified from (Hillestad et al. 1975)
The soils of the barrier islands were derived primarily from quartz sands, which are highly resistant to decomposition by chemical and physical weathering processes. They have resisted environmental degradation for 35-50,000 years and still closely resemble their parent materials. The coarse-to-fine sandy soils of the islands are commonly placed in the Regosol soil group (i.e., soils with a poorly differentiated profile).

Atmospheric fallout (Art 1974), high tidewaters and terrestrial birds and mammals that feed on marine organisms bring nutrients to the islands from the sea. These same agents also return nutrients to the sea, and the net gain, if any, is small. Nutrients are at a premium on the islands.

The dominant feature of island soils is their high permeability, which results in low water-holding capacity and rapid leaching. The lack of cation (Na, K, Ca, etc.) adsorption sites on the quartz crystals produces soils with a low cation exchange capacity (CEC) and, consequently, an inability to retain essential plant nutrients. Nutrients, therefore, are vigorously recycled, and at any given time most cations will be complexed in plants or organic humus. Microbial decomposition of organic matter releases the nutrients and they again are quickly tied up by vegetation.

Barrier Islands

Coastal barriers are geologically recent depositional sand bodies that are highly variable in shape, size, and their response to natural processes and human alterations (White et al. 1998). They may stretch many kilometers in length and contain high sand dunes--such as the Outer Banks of North Carolina – or they may be small and isolated islands, so low in relief that they are routinely overwashed by spring tides and minor storms. Their dynamic nature means coastal barriers are constantly shifting and being modified by winds and waves, but scientific field investigations over the past several decades are revealing some disturbing trends.

Earthquakes – from (South Carolina Seismic Network 2003)

The seismic history of the southeastern United States is dominated by the 1886 earthquake that occurred in the Coastal Plain near Charleston, South Carolina. It was one of the largest historic earthquakes in eastern North America, and by far the largest earthquake in the southeastern United States. A major shock, occurred August 31, 1886 at approximately 9:50 p.m. and lasted less than one minute, but resulted in about sixty deaths and extensive damage to the city of Charleston. Because the event took place before seismological instrumentation, estimates of its location and size must come from observations of the damage and effects caused by the earthquake. Most of what we know of the resulting damage comes from a comprehensive report by C.E. Dutton of the U.S. Geological Survey published in 1889. The meizoseismal area (area of maximum damage) of the 1886 earthquake is an elliptical area roughly 20 by 30 miles trending northeast between Charleston and Jedburg and including Summerville and roughly centered at Middleton Place.

The 1886 earthquake was followed by a series of aftershocks. Of 435 or more earthquakes reported to have taken place in South Carolina between 1754 and 1975, more than 300 were aftershocks that occurred in the first 35 years following 1886. The 1886 earthquake and its aftershocks dominate the seismic record of the southeast.

The historic record suggests the Charleston-Summerville area had a continuum of low level seismicity prior to 1886, and low-level activity continues in the same area today.

Weather & Climate – *modified from (White et al. 1998)*

Southeastern climates are humid and warm-temperate to subtropical. Major variation in climate occurs with change in latitude and elevation. Longitude has a more subtle influence on climate than latitude, as a result of maritime influence to the south and east and continental influences to the north and west.

Latitudinal gradients in temperature are steeper in winter than in summer, producing a strong geographic pattern in freeze-free periods and cold temperatures. The gradient in average minimum January temperature spans 22°C, whereas the gradient in average maximum July temperature spans only 4°C (Ruffner 1985, Martin and Boyce 1993). The freeze-free period decreases northward, from 365 days in the Florida Keys, which experienced freezing temperatures in fewer than half of the years on record, to 180 days in Arkansas and 150 days in northern Virginia. The freeze-free period also decreases with elevation, to 110 days at the highest elevations in the southern Appalachians. Canadian air masses bring the coldest winter temperatures, penetrating the Southeast from the continental interior and generally producing decreasing minimum temperatures westward at a given latitude. Annual snowfall shows the same steep gradients as cold winter temperatures, increasing from zero in south Florida to over 100 centimeters northward and to over 200 centimeters in the high mountains.

Annual precipitation averages 110-140 centimeters over much of the area, with a slight decrease northward to about 100 centimeters. Excluding the high mountains, the highest annual precipitation occurs along the Gulf of Mexico coast and in south Florida (140-160 centimeters). Annual precipitation increases to 200 centimeters where elevations surpass about 1,700 meters. The highest values are not, however, at the extreme elevations but are affected by the position of the mountain front relative to precipitation sources. The first high mountains encountered by moist air masses from the Gulf of Mexico coast and the Atlantic are those at the southern edge of the Blue Ridge near the joint boundaries of the region of North Carolina, South Carolina, and Georgia. This region has the Southeast's highest precipitation (as much as 250 centimeters) and the highest rainfall in the United States east of the Pacific Northwest.

Precipitation occurs throughout the year but is generally lowest in fall and highest in summer, when convective thunderstorms develop. Thunderstorms in Florida occur an average of 80-130 days annually, in the Gulf Coastal Plain 80-100 days annually; the number of thunderstorms decreases northward, occurring an average of 40-60 days a year in Kentucky, Virginia, and interior regions.

By combining climate and physiography, McNab and Avers (1994) classified the Southeast into 2 domains (humid temperate and tropical), 3 divisions (humid temperate, hot continental; humid temperate, subtropical; and humid tropical, savanna), 9 provinces, and 28 sections, the latter representing distinctive landscape types.

Park Resources

Individuals & Populations

Phenotypic Polymorphism / Diversity
Phenotypic polymorphism refers to the occurrence in a population (or among populations) of several phenotypic forms associated (but not entirely driven by) genes. Phenotypic polymorphism can be caused by environmental influences or biological interactions (Summers et al. 2003), and can result in within-population differences in morphology, feeding behavior, and reproductive behavior (Robinson et al. 1993, Kriegsfeld et al. 2000, Plague et al. 2001). Phenotypic polymorphism can itself be an adaptive trait to withstand environmental uncertainty (Yoshimura and Clark 1991). An example of phenotypic polymorphism within the SECN is temperature-driven gender determination in sea turtles.

Species, Assemblages, & Communities

Reptiles & Amphibians – *modified from White et al. (1998)*
Reptiles and amphibians are present in virtually all natural habitats in the Southeast. All the turtle species nest on land, some aquatic turtles and snakes hibernate on land, and dozens of species of southeastern frogs and salamanders are terrestrial as adults but require wetlands for breeding and development of young. Also, terrestrial corridors among aquatic habitats are essential for reptile and amphibian dispersal during unfavorable periods such as drought.

The Southeast has the highest regional total (130 species) of amphibians in the United States (Echternacht and Harris 1993), including 38 species of frogs and toads (12 of these are endemic to the Southeast) and 92 species of salamanders (45 of which are endemic to the Southeast). The southern Appalachians are a world center of diversity for salamanders and have 68 species of a unique group of lungless salamanders that evolved in this region of well-oxygenated streams and high rainfall. The Southeast has 6 species of large, fully aquatic salamanders and the Coastal Plain has 32 species of frogs and toads, of which 11 are endemic.

There are 52 species of snakes in the Southeast, of which 11 are endemic (Conant and Collins 1991, Echternacht and Harris 1993). Of the 91 species of lizards native to the United States, 21 occur in the Southeast, and 6 of these are endemic. The Southeast has 36 species of turtles, 13 of which are endemic; the Coastal Plain possesses North America's highest diversity in this group. One of the two greatest concentrations of freshwater turtle species in the world (the other is in Asia) is in the Mobile River basin (Iverson 1992, Lydeard and Mayden 1995).

The greatest threat to reptiles and amphibians comes from habitat loss and changes in water quality. Numerous examples can be given of population declines in individual wetlands as a consequence of human activities. Drainage and destruction of temporary ponds have resulted in the reduction of striped newts in Georgia (Dodd, Jr. 1995a), the extirpation of the flatwoods salamander from a portion of its range, and apparent declines of gopher frogs in Alabama and Mississippi (Dodd, Jr. 1995a).

Species that are adapted to terrestrial habitats have also suffered. Of the 242 native reptiles and amphibians in the Southeast, 170 (74 amphibians, 96 reptiles) are native to longleaf pine-wire-grass ecosystems (Dodd, Jr. 1995a). The near loss of this natural community, through timbering, development, and fire suppression, has had a significant, though largely unquantified, effect on reptiles and amphibians.

Highway deaths also deplete the numbers of many species of reptiles and amphibians that travel overland. A 2-meters-long indigo snake, for example, does not move fast enough to safely get across today's highways.

Some ecologists have reported declines in amphibian populations and related these to specific threats, such as acid rain, destruction of the ozone layer, global warming, or other forms of nonpoint pollution (Blaustein 1994). It is unclear if any of these factors are responsible for amphibian declines in some regions (Pechmann et al. 1991, Pechmann and Wilbur 1994), but habitat destruction is the primary threat to most species of reptiles and amphibians in this country and probably in most countries in the world today. Timber harvest, for example, dramatically reduces amphibian populations in the southern Appalachians (Petranka et al. 1993). Habitat destruction may take more subtle forms, though, and what may appear to be protected and pristine habitat may actually be experiencing degradation because of changes in hydrology, pollution, herbicide and pesticide runoff, the introduction of competitive nonindigenous species, the introduction of disease organisms, or the loss of important breeding sites such as temporary ponds (Blaustein 1994, Dodd, Jr. 1995b).

Insufficient knowledge of the distribution and ecology of native reptiles and amphibians is a major shortcoming in any national effort to detect change and avoid loss in this group. An example of the difficulty that ecologists face in confirming the presence of herpetofauna is apparent from studies by investigators at the Savannah River Ecology Laboratory and from studies by other investigators on the Savannah River Site in South Carolina. This site is the largest tract of land (750 square kilometers) in North America with high herpetofaunal species diversity and a long-term record of intensive ecological research and survey. Since the 1950's, herpetologists have collected data on more than a million individual reptile and amphibian specimens representing more than 100 species (Gibbons and Semlitsch 1991). Nonetheless, despite intensive surveys, the presence of new species has been verified on the Savannah River Site at a rate of more than five species per decade.

Marine Turtles – modified from Dodd (1995a)

Five species of marine turtles frequent the beaches and offshore waters of the southeastern United States: loggerhead (*Caretta caretta*), green (*Chelonia mydas*), Kemp's ridley (*Lepidochelys kempii*), leatherback (*Dermochelys coriacea*), and hawksbill (*Eretmochelys imbricata*). All five are reported to nest, but only the loggerhead and green turtle do so in substantial numbers. Most nesting occurs from southern North Carolina to the middle west coast of Florida, but scattered nesting occurs from Virginia through southern Texas. The beaches of Florida, particularly in Brevard and Indian River counties, host what may be the world's largest population of loggerheads.

Marine turtles, especially juveniles and subadults, use lagoons, estuaries, and bays as feeding grounds. Areas of particular importance include Chesapeake Bay, Virginia (for loggerheads and Kemp's ridleys); Pamlico Sound, North Carolina (for loggerheads); and Mosquito Lagoon, Florida, and Laguna Madre, Texas (for greens). Offshore waters also support important feeding grounds such as Florida Bay and the Cedar Keys, Florida (for green turtles), and the mouth of the Mississippi River and the northeast Gulf of Mexico (for Kemp's ridleys). Offshore reefs provide feeding and resting habitat (for loggerheads, greens, and hawksbills), and offshore currents, especially the Gulf Stream, are important migratory corridors (for all species, but especially leatherbacks).

Most marine turtles spend only part of their lives in U.S. waters. For example, hatchling loggerheads ride oceanic currents and gyres (giant circular oceanic surface currents) for many years before returning to feed as subadults in southeastern lagoons. They travel as far as Europe and the Azores, and even enter the Mediterranean Sea, where they are susceptible to longline fishing mortality. Adult loggerheads may leave U.S. waters after nesting and spend years in feeding grounds in the Bahamas and Cuba before returning. Nearly the entire world population of Kemp's ridleys uses a single Mexican beach for nesting, although juveniles and subadults, in particular, spend much time in U.S. offshore waters.

The biological characteristics that make sea turtles difficult to conserve and manage include a long life span, delayed sexual maturity, differential use of habitats both among species and life stages, adult migratory travel, high egg and juvenile mortality, concentrated nesting, and vast areal dispersal of young and subadults. Genetic analyses have confirmed that females of most species return to their natal beaches to nest (Bowen et al. 1992, Bowen et al. 1993). Nesting assemblages contain unique genetic markers showing a tendency toward isolation from other assemblages (Bowen et al. 1993); thus, Florida green turtles are genetically different from green turtles nesting in Costa Rica and Brazil (Bowen et al. 1992). Nesting on warm sandy beaches puts the turtles in direct conflict with human beach use, and their use of rich offshore waters subjects them to mortality from commercial fisheries (National Research Council 1990).

Marine turtles have suffered catastrophic declines since European discovery of the New World (National Research Council 1990). In a relatively short time, the huge nesting assemblages in the Cayman Islands, Jamaica, and Bermuda were decimated. In the United States, commercial turtle fisheries once operated in south Texas (Doughty 1984), Cedar Keys, Florida Keys, and Mosquito Lagoon; these fisheries collapsed from overexploitation of the mostly juvenile green turtle populations. Today, marine turtle populations are threatened worldwide and are under intense pressure in the Caribbean basin and Gulf of Mexico, including Cuba, Mexico, Hispaniola, the Bahamas, and Nicaragua. Subadult loggerheads are captured extensively in the eastern Atlantic Ocean and Mediterranean Sea. Thus, marine turtles that hatch or nest on U.S. beaches or migrate to U.S. waters are under threats far from U.S. jurisdiction. Marine turtles can be conserved only through international efforts and cooperation.

Information on the status and trends of southeastern marine turtle populations comes from a variety of sources, including old fishery records, anecdotal accounts of abundance, beach surveys for nests and females, and trawl and aerial surveys for turtles offshore. Surveys for marine turtles are particularly difficult because most of their lives are spent in habitats that are not easily surveyed. Hence, most status and trends information comes from counting females and nests. Few systematic long-term (more than 10-20 years) surveys have been conducted; the most notable are the nesting surveys at Cumberland Island and adjacent barrier islands in Georgia (T.H. Richardson, University of Georgia, unpublished data), Canaveral National Seashore (1988 – present), and beaches south of Melbourne in Brevard County, Florida (Ehrhart et al. 1993). Beach monitoring is fairly widespread in many areas of the Southeast, but coverage varies considerably among beaches and field crews. The only long-term sampling of lagoonal or bay populations occurs at Mosquito Lagoon and Chesapeake Bay, although short-duration surveys have sampled Florida Bay, Pamlico Sound, and Laguna Madre. Trawl surveys of inlets and ship channels and aerial surveys of offshore waters have been undertaken periodically.

Sea turtles are threatened by beach development, light pollution, ocean dumping, incidental take in trawl and longline fisheries, disease (especially fibropapillomas), and many other variables. Because sea turtles are long-lived species, trends are difficult to monitor. Present methods of beach monitoring are extremely labor-intensive, expensive, and biased toward one segment of the population. Very little is known about marine turtle life-history and habitat requirements away from nesting beaches, and virtually nothing is known about male turtles. Because the effectiveness of measures aimed at protecting turtles may not be seen for decades, known conservation strategies should be favored over unproven mitigation schemes. Acquiring nesting habitat should be encouraged. One of the most important management measures to protect sea turtles, especially of the juvenile and subadult size class, in the southeastern United States, Caribbean, and western Atlantic Ocean is the use of TEDs to minimize drowning in commercial fisheries. Mature females should also be protected because of their importance to future reproduction. Researchers need to identify migratory routes, feeding and developmental habitat, and ways to minimize adverse impacts during all life-history stages.

Gopher Tortoise – modified from Neal (1990)

An understanding of the reasons behind the threatened status of gopher tortoises (*Gopherus polyphemus*) is perhaps the most essential step in developing this recovery plan. The gopher tortoise, historically and currently, is a component of xeric plant communities originally identified mostly by the occurrence of longleaf pine. The changes altering the original longleaf pine communities also changed the ecosystem of the gopher tortoise. This species was an animal of these forests, and to the extent maintenance of the listed population is possible, that goal is inextricably tied to forestland conditions.

Before the arrival of European colonists in the New World, the longleaf pine was the principal tree species on southeastern coastal plain upland soils. Croker (1987) cites 60 million acres in the original stands which he concludes are now reduced to about 4 million acres. After the red and white pine forests of New England and the Great Lake States were cut, lumbermen turned to the virgin longleaf stands, the mining of which peaked in 1909 (Croker 1987). Power skidders and railroad logging supported these final assaults.

Second growth longleaf pine stands came from the ruins of timber mining operations, but these second forests constituted a small fraction of the area of virgin stands. Because of planting difficulties with the longleaf pine, these droughty sites were often planted in slash (*Pinus elliottii*) and loblolly (*P. taeda*) pines. This practice, along with excessive burning intervals and intensive site preparation methods, continues on soils which originally supported longleaf pine.

Artificial planting of longleaf is now successful and many foresters are rediscovering the valuable traits of longleaf pine, including the fact that it can be successfully regenerated naturally through a shelterwood system of cutting combined with burning just in advance of an adequate seed fall. The U.S. Forest Service recently has adopted a practice of regenerating only longleaf pines on longleaf sites in the DeSoto National Forest. However, the agency's preferred method is by planting. Most private landowners continue to regenerate longleaf pine sites to off-site species.

The original longleaf pine community burned and reseeded naturally. It contained trees of many ages and a diverse ground cover with much edge, which would be of particular importance to the gopher tortoise. Landers and Speake (1980) found better gopher tortoise densities in longleaf pine-scrub oak stands that were thinned and burned every 2-4 years. Slash pine plantations, with a similar system of thinning and burning, had sparser population densities. While it is apparent that gopher tortoises can be maintained under a modified (heavily thinned, frequently burned) plantation system of management, Landers and Buckner (1981) showed that gopher tortoise densities are significantly greater (32 percent) in more naturally managed stands of longleaf.

The natural longleaf pine community and its associated biological diversity represent optimal forest habitat for the gopher tortoise. This community occurred in pure stands, constantly trending toward small even-aged groups of a few hundred square feet (Chapman 1909). Larger even-aged patches and strips were found following blowdowns from severe weather. These were often interspersed with patches or single survivors, creating open glades and a patchiness which favored the gopher tortoise. Management practices which alter this system include: clearcuts of large blocks (including the crowded planting of off—site species), diversity—diminishing soil churning activities that often accompany even—aged timber manag9m~nt, and prolonged burning intervals. Timber practices that most nearly mirror the natural system, such as a shelterwood regeneration system with frequent burning and natural regeneration, improve the soil and herbaceous cover condition to optimally support the gopher tortoise.

Longleaf pine trees, as well as fire-dependent annuals and perennials, originally existed in a summer burning cycle which has long since been interrupted. The change in fire frequency and timing may be the single most important factor influencing other alterations which have changed the original xeric communities. For example, it has been a common practice to remove most of the longleaf pines from these dry ridges and then to exclude fire (or at least fail to burn). This allows eventual occupancy by poor site oaks (*Quercus laevis*, *Q. incana*, *Q. marilandica*, and *Q. margaretta*) and woody shrubs such as yaupon (*Ilex vomitoria*) and gallberry (*I. glabra*). When the leaf litter from oaks becomes a thick mat, it retards fires that would otherwise be carried by longleaf pine needles and the common grass associates under the open longleaf pine canopy. Fire exclusion allows the oaks to mature and shade out herbaceous ground cover needed by gopher tortoises. This situation is not uncommon throughout the range of the gopher tortoise. Landers and Speake (1980) provided substantial evidence that these altered sites originally were good gopher tortoise habitat but now support the fewest gopher tortoises.

Additional threats to gopher tortoise populations include mortality from predators (including humans), road mortality, and disease transmission.

Diamondback Terrapin – modified from Golder et al. (2004) and Diamondback Terrapins eGroup (2004)
During the late 1800s' into the late '20s of the previous century, diamondback terrapin (*Malaclemys terrapin*) populations were greatly depleted in the wild due to harvesting for the gourmet food market. Fortunately, several states acted in a timely fashion to protect them from complete annihilation, and diamondback numbers soon recovered in the wild. Their recovery was so dramatic and successful, that the states lifted their protective status and commercial trade in diamondbacks was once again allowed. Fortunately, terrapin meat was no longer in demand and diamondbacks were allowed to thrive in their natural habitats until recently. Now, diamondback populations are no longer threatened by just commercial harvesting but are faced with having to contend with urban progress as well as the effects of commercial crabbing. Habitat destruction, road kills, mass drowning in crab traps, and pollution are now threatening the status of diamondback populations throughout their range. A number of states have the species listed as either protected or of special concern. Among them is Florida (home to 4 different diamondback subspecies), which has again adopted protective measures to shut down commercial exploitation of terrapins.

Terrapins are found exclusively in brackish water habitats with the exception of the egg laying season, when gravid females venture out of the water and marshes to lay their eggs on dry land. Unfortunately, many nest sites have been destroyed by coastal development in the recent years. It is common knowledge that coastal property is prime real estate, and the continued encroachment of human development into diamondback habitat is not only depleting their habitat, but causing other lethal problems as well. Female terrapins that are no longer able to find nesting sites on developed barrier beach islands are forced to seek out alternative sites to lay their eggs. In the process, many gravid diamondback females are lost to traffic mortality annually, resulting in not only the deaths of the nesting females, but also the loss of viable diamondback eggs. A species cannot continue to survive without adequately replenishing its population.

Another factor that is having devastating effect on diamondback populations is the drowning of terrapins in commercial and recreational crab traps. Conservative numbers suggest that tens of thousands of diamondbacks drown in these traps annually. Terrapins breath oxygen from the air, unlike the blue crabs that the majority of these traps were designed to catch. While road kills have been instrumental in reducing the number of females within a certain population, crab traps have the same effect on juveniles and males of the species. Female terrapins are twice the size of males and thus have a tendency not to be able to fit into the traps.

Although commercial harvesting is done at a considerably smaller level than earlier this century, it too is having a negative effect on wild terrapin populations. Although its effects are not as serious as those caused by the other factors mentioned earlier, coupled with these other factors, commercial harvesting can no longer be categorized as "sustainable use." Most collecting is now done for both the food industry and the pet trade. Once again, it is the adult females that are most valued to the food trade and this raises severe concerns as to whether the species can adequately replenish itself with continued indiscriminate harvesting.

The diamondback terrapin (*Malaclemys terrapin*) has been considered a status review species by the US Fish and Wildlife Service for over a decade. In part its conservation status has remained unchanged because the USFWS watches state designations and most states have little information concerning current population trends. Issues are further complicated by the fact that many states cover terrapin regulations under fisheries units, while state wildlife agencies typically oversee conservation status listings. Several key issues face diamondback terrapins. These include a commercial harvest, unregulated and illegal traffic of animals for food markets, loss of nesting beaches through shoreline erosion and bulkheads constructed to prevent erosion, road mortality of nesting females, increased egg predation by growing populations of raccoons which are supplemented by garbage associated with coastal development, and fatal collisions with boats and jet skis. One of the most serious problems is the drowning of terrapins accidentally captured in crab pots. New Jersey and Maryland require terrapin excluders on all recreational crab pots. In several states similar programs are now under consideration.

Mammals – *modified from White et al. (1998)*
Terrestrial and freshwater habitats in the Southeast are home to 101 mammal species (Echternacht and Harris 1993). Of these, 5 are extirpated, all of them ecologically important large carnivores or grazers: jaguar, ocelot, gray

wolf, elk, and bison (Echternacht and Harris 1993). Two other large carnivores are on the verge of extinction: the Florida panther, the only remaining subspecies of mountain lion in the eastern United States, and the red wolf.

Endemic species represent a relatively small percentage of the mammals. According to Echternacht and Harris (1993), eight small mammal species are endemic to the Coastal Plain province of the Southeast: southeastern pocket gopher, colonial pocket gopher, Sherman's pocket gopher, Cumberland Island pocket gopher, oldfield mouse, Florida mouse, Perdido Key beach mouse, and round-tailed muskrat. The region also has eight species of introduced mammals, four of which have many adverse effects on native communities: coyote, pig (feral domesticated pigs and wild boar) in the mountains and Coastal Plain, and nutria and horse in the Coastal Plain. Beavers were extirpated in the Southeast but have become reestablished in the last 20 years. Although beavers were historically important in the maintenance of habitat diversity, beavers of today inhabit landscapes with reduced predation and where the remnant habitats may themselves be vulnerable to loss from flooding.

There are 22 federally listed mammals in the Southeast: eastern mountain lion and the Florida panther, Key deer, gray wolf, red wolf, Louisiana black bear, 4 species of bats, 9 small mammal species restricted to the Coastal Plain in Florida or Alabama, a shrew restricted to Virginia and North Carolina, and 2 species of flying squirrels restricted to the mountains (Lee et al. 1982, Humphrey 1992). The eastern mountain lion and the gray wolf are already extirpated in the Southeast. In the following sections we discuss these and other species representative of trends in southeastern mammals.

Small Mammals – from White et al. (1998)

Small mammal species that are most at risk in the Southeast have narrow distributions. Most of the threats to these species come from development and subsequent loss of habitat. In isolated communities, such as beach habitats, feral cats represent a significant threat. Shrews and other insectivorous mammals suffer from the concentrated effects of residual pesticides. Fleming and Holler (1989) described ongoing efforts to reintroduce the endangered Perdido Key beach mouse to a site in Gulf Islands National Seashore.

The future of the fox squirrel is linked to that of its habitat, the longleaf pine savannah. A long-lived species with low reproductive rates, the fox squirrel has not been well studied or understood, but timbering, fire suppression, and development are all limiting its range and reducing its population sizes.

Bats – from White et al. (1998)

Of the 39 bat species listed for the United States, 17 occur in the Southeast (Di Silvestro 1989). Widespread pesticide use, resulting in poisoning as well as loss of food sources, is responsible for significant declines in bat populations since the 1960's (Di Silvestro 1989, Humphrey 1992, Drobney and Clawson 1995). This threat has diminished with regulations on pesticide use. The greatest threat to bats now comes from habitat destruction and disturbance. Few caves meet the temperature and humidity requirements bats need for hibernation, and these caves are occupied by large numbers of bats, making these bats particularly vulnerable to disturbance. The slow rate of reproduction among bats (often only one offspring per year) means that a population can be quickly destroyed, with little opportunity for recovery (Di Silvestro 1989).

The Indiana bat ranges over a huge area of the eastern United States, but the winter habitat for 85% of the species is limited to just seven caves, with over half of the population using just two caves (Di Silvestro 1989). Human disturbance has caused numbers of this species to drop from 330,000 to 49,000 in Kentucky alone (Di Silvestro 1989). Nationally, the decline in the Indiana bat population has reached 22% in the past 10 years (Drobney and Clawson 1995). Missouri has experienced the greatest decline (34%), whereas bat numbers in Indiana have somewhat increased and Kentucky's population is now stable.

The gray bat has suffered a similar fate. Guano collection during the Civil War caused heavy losses initially because of disturbances to nursery caves and habitats, but the gray bat recovered, only to be decimated by the popularity of cave exploration in the 1960's and 1970's. Between 1970 and 1976 the population of some colonies dropped more than 50%. Though only a handful of caves are suitable for the gray bat, this species is showing signs of recovery, largely due to the protection of four critical caves (Di Silvestro 1989).

Manatees (Lefebvre and O'Shea 1995)
The endangered Florida manatee (*Trichechus manatus latirostris*) is a survivor. It is one of only three living species of manatees which, along with their closest living relative, the dugong (*Dugong dugon*), make up the Order Sirenia. This taxonomic distinctiveness reflects their evolutionary and genetic uniqueness. Sirenians are the only herbivorous marine mammals; manatees feed on seagrasses; freshwater plants, including nuisance species such as hydrilla and water hyacinth; and even some shoreline vegetation. Because manatees depend on marine, estuarine, and freshwater ecosystems, our efforts to protect them necessitate protection of aquatic resources.

Species recovery criteria for the Florida manatee are three-fold: the population trend must be stable or increasing; mortality must be stable or declining; and threats to manatee habitat must be under control (U.S. Fish and Wildlife Service 1989). Better population and life-history data suggest a greater potential for increase and higher numbers than previously recognized and strong steps taken by local, state, and federal governments are increasing the number and area of sanctuaries and slow boat-speed zones. These steps may reduce mortality if they are continued and expanded, allowing the population to recover more quickly.

Management has focused on ways to reduce human-related mortality. Of greatest concern has been an increase over the years in the number of human-caused deaths, particularly those caused by collisions with boats. Boat strikes account for 78% of human-related manatee mortality and 25% of all documented deaths (O'Shea et al. 1995). A moderate reduction in the number of boat-related deaths in the last 2 years caused optimism; however, watercraft collisions accounted for 49 manatee deaths in 1994, almost matching the record number of 51 in 1991.

Habitat threats are far from under control, however. Florida has one of the fastest-growing human populations in the nation, with an estimated net gain of close to 1,000 people per day (Fernald et al. 1992). Much growth has occurred along the coast, with inevitable consequences for coastal habitats. For example, about a third of the 600,000 ha (1.5 million acres) of seagrass meadows present in coastal Florida in the 1940's no longer exist (Lewis, III 1987). One of the most important regions for manatees on the Atlantic coast is the Indian River Lagoon. Over the past 20 years, losses of submerged aquatic vegetation in some areas of the lagoon have exceeded 95% (Busby and Virnstein 1993). Submerged freshwater plants have also been affected adversely by increases in turbidity and nutrients.

Debris, particularly monofilament line, plastics, and unattended fishing nets and ropes, directly threatens manatees, who may ingest or become entangled in these materials (Beck and Barros 1991). Manatees are also vulnerable to natural and human-caused catastrophes, such as disease and oil spills, particularly when the animals are concentrated at winter aggregation sites.

Deer – modified from White et al. (1998)
White-tailed deer populations have fluctuated dramatically with changing human influence and land use. We can identify four periods of contrasting trends and influence on native ecosystems. Before 1500, deer populations were moderate in size--Native Americans hunted deer extensively, and large native predators of deer were also present. Between 1500 and 1800, deer populations probably increased in some areas and decreased in others. Increases occurred because of reduced hunting by Native Americans and the increase in old-field habitats as Native American farms and villages were abandoned after Europeans displaced the native populations. Decreases were the result of exploitive hunting for trade by Native Americans and European colonists. Between 1800 and 1930 deer populations were reduced to near extirpation in many areas because of increased hunting, widespread agricultural clearing, and also other causes such as draining of wetlands. Since 1930 deer populations have rebounded vigorously because of farm abandonment, lower hunting pressure, and the near-absence of natural deer predators. Deer populations are still increasing in the Southeast and in some areas are drastically altering the composition and density of understory stems in forests. Deer are a major issue in forest and conservation management.

Birds – modified from White et al. (1998)
The Southeast originally had 237 native species of birds, none of which were strictly endemic to the region (Echternacht and Harris 1993). Three species are nearly restricted to the Southeast: Bachman's warbler (which may be the rarest vertebrate in the region), Swainson's warbler, and the brown-headed nuthatch. Twenty-six percent of

the total (61 species) is associated with water. Of these, 19 are large wading bird species, a group for which the Southeast has the continent's highest total. The greatest species richness of birds occurs in the coastal wetlands. Thirty-one species (13.4%) are restricted to the high mountains. Echternacht and Harris (1993) estimated that there are 17 established nonindigenous bird species in the Southeast, but they warned that the number may be an underestimate, considering that other species have been released in the area.

Land clearing and hunting were responsible for the extinction of two bird species in the Southeast: the passenger pigeon (last reported in the wild in 1899) and the Carolina parakeet (last reported in the wild in 1913). Passenger pigeons were hunted for their market value whereas Carolina parakeets, birds of old wetland forests, were hunted to protect fruit crops.

Three species have been extirpated from the Southeast: ivory-billed woodpecker (last seen in the 1950's and thought to persist in Cuba), which was dependent on large-cavity trees in extensive and old riparian forests; and the Zenaida dove and the Key West quail-dove, which were rare Caribbean species restricted to Florida-- the reason for their extirpation is not known (Echternacht and Harris 1993). An additional subspecies, the dusky seaside sparrow, became extinct because of poor fire management of its marsh habitat in coastal northern Florida.

Fourteen species and subspecies of birds are federally listed, of which 12 are Coastal Plain species: crested caracara, Mississippi sandhill crane, Florida scrub-jay, brown pelican, piping plover, Cape Sable seaside sparrow, dusky seaside sparrow (now extinct), wood stork, least tern, Bachman's warbler, ivory-billed woodpecker, and red-cockaded woodpecker. The fate of these species is largely tied to habitat loss, including reductions in longleaf pine savannah, Florida scrub, wetlands, and beach communities. Two other federally listed species, the bald eagle and the peregrine falcon, were formerly wide-ranging species sensitive to pesticides; these species are now recovering.

The Southeast is important not only for summer breeding populations but also for birds that winter in the Southeast and for birds that migrate farther distances (for example, to the Caribbean and Central and South America) after passing through the South in spring and fall. Coastal habitats, maritime forests, and longleaf pine savannah are all important to migrating species. Threats to bird species include land-use changes, forest fragmentation (which often results in increased nest predation and cowbird parasitism), tropical deforestation (for Neotropical migrants), elimination of wetlands, and coastal development.

Critical information for the conservation of bird species includes understanding the relationship between reproductive success and habitat size and quality. Hunter (draft report) stated that to create populations that will endure and that will generate excess individuals to colonize new sites, some bird species (for example, the ivory-billed woodpecker) require 2,000 to 40,000 hectares of unbroken habitat. Further, we have to understand the relation between reproductive success and such microhabitat variables as forest-age structure. Hunter also reported that species that require large areas can act as umbrella species for species with smaller area requirements. If we understand the habitat area each bird species needs, it will help us determine optimum block sizes and rotations for harvested forests. The need for large habitat areas is another argument for reforestation of marginal farmlands and the retention of wetlands. Because the southeastern landscape is so heavily in private ownership, land used for agriculture and forestry must play a large role in the survival of bird species diversity. Erwin (1995) suggested that recent increases in great blue heron populations resulted from this bird's practice of feeding in aquaculture ponds. Finally, regional monitoring of bird populations is essential because of geographic movements of species. For example, white ibis and wood stork populations have declined in south Florida but are stable in the Southeast as a whole because of population shifts northward to northern Florida, Georgia, and the Carolinas (Erwin 1995).

Red Cockaded Woodpeckers – modified from Costa and Walker (1995)

The red-cockaded woodpecker (RCW; *Picoides borealis*) is a territorial, nonmigratory, cooperative breeding species (Lennartz et al. 1987). Ecological requirements include habitat for relatively large home ranges (34 to about 200 ha or 84 to about 500 acres; Connor and Rudolph 1991); old pine trees with red-heart disease for nesting and roosting (Jackson and Schardien 1986); and open, parklike forested landscapes for population expansion, dispersal (Connor and Rudolph 1991), and necessary social interactions.

Historically, the southern pine ecosystems, contiguous across large areas and kept open with recurring fire (Christensen 1981), provided ideal conditions for a nearly continuous distribution of RCWs throughout the South. Within this extensive ecosystem red-cockaded woodpeckers were the only species to excavate cavities in living pine trees, thereby providing essential cavities for other cavity-nesting birds and mammals, as well as some reptiles, amphibians, and invertebrates (Kappes 1993). The loss of open pine habitat since European settlement precipitated dramatic declines in the bird's population and led to its being listed as endangered in 1970.

The historical range of this species covered southeast Virginia to east Texas and north to portions of Tennessee, Kentucky, southeast Missouri, and eastern Oklahoma. The range included the entire longleaf pine ecosystem, but the birds also inhabited open shortleaf, loblolly, and Virginia pine forests, especially in the Ozark-Ouachita Highlands and the southern tip of the Appalachian Highlands.

The decline of the red-cockaded woodpecker coincided with the loss of the longleaf ecosystem. As forests were cleared, birds were isolated in forest tracts where unmerchantable trees were left. Aerial and ground photographs from the 1930's show that scattered medium to large trees (0.4-2 per ha or 1-5 per acre) were left in many stands. The culled trees (undoubtedly including red-cockaded woodpecker cavity trees) provided residual nesting and foraging habitat for the birds. In some places these trees remain and are used by red-cockaded woodpeckers today.

Since the 1950's, on lands managed for forest products, the forest structure and composition changed in conjunction with clear cutting, short timber rotations, conversion of longleaf stands to other pine species, and "clean" forestry practices (removal of cavity, diseased, or defective trees). These practices eliminated much of the remaining red-cockaded woodpecker habitat. Additionally, aggressive fire suppression promoted the development of a hardwood midstory in pine forests. The adverse impacts of a dense midstory on RCW populations are well-documented (Connor and Rudolph 1989, Costa and Escano 1989).

The Red-cockaded Woodpecker Recovery Plan (U.S. Fish and Wildlife Service 1985) specifies that range-wide recovery will be achieved when 15 viable populations are established and protected by adequate habitat management programs. The recovery populations are to be distributed across the major physiographic provinces and within the major forest types that can be managed to sustain viable populations. Each recovery population will likely require 400 breeding pairs (or 500 active clusters, as some clusters are occupied by single birds or contain non-breeding groups) to ensure long-term population viability (Reed et al. 1993). At a density of 1 group/80-120 ha (200-300 acres; USFWS 1985; USFS 1993), landscapes of at least 40,000 ha (100,000 acres) will be needed to support viable populations. Most forested pine areas large enough to supply this habitat are on public, mostly federal, lands.

With two exceptions (Hooper et al. 1991); USFS, Apalachicola National Forest, FL, unpublished data), there is no evidence that red-cockaded woodpecker populations can expand to viable levels without considerable human intervention. Conversely, numerous population extirpations have been documented (Baker 1983, Costa and Escano 1989). Ensuring the survival of the species, even in the short term (50 years), will require landscape-scale habitat and population management to provide the forest structure and composition needed for nesting and foraging habitat and population expansion; and to manage limiting factors (primarily a lack of suitable cavity trees, cavity competition, and demographic isolation) that can extirpate small populations. Both strategies are part of management guidelines drafted by several federal land stewards (U.S. Forest Service 1993, U.S. Army 1994, U.S. Fish and Wildlife Service 1994).

These ecosystem management plans promote practices that minimize landscape fragmentation, retain suitable numbers of potential cavity trees well distributed throughout the landscape, and restore the original forest cover by planting the appropriate pine species. They recommend the use of growing-season fires to control hardwoods, create open forest conditions, and begin to restore the understory plant communities of the pine ecosystems. Stabilization and growth of small high-risk populations will be aided by creating artificial red-cockaded woodpecker cavities (Copeyon 1990) and translocating juvenile birds from stable larger populations into small ones (Rudolph et al. 1992). Technologies that minimize or eliminate predation and competition problems are available (Carter, III et al. 1989).

Shorebirds – Modified from Harrington (1995)

The North American group of shorebirds includes 48 kinds of sandpipers, plovers, and their allies, many of which live for most of the year in coastal marine habitats; others live principally in non-marine habitats including grasslands, freshwater wetlands, and even second-growth woodlands. Most North American shorebirds are highly migratory, while others are weakly migratory, or even nonmigratory in some parts of their range. Here we discuss shorebirds east of the 105th meridian (roughly east of the Rocky Mountains). Historically, populations of many North American species were dramatically reduced by excessive gunning (Forbush 1912). Most populations recovered after the passage of the Migratory Bird Treaty Act of 1918, although some species never recovered and others have declined again.

High proportions of entire populations of shorebirds migrate by visiting one or a small number of "staging sites," areas where the birds accumulate fat to provide fuel before continuing with their long-distance, nonstop flights to the next site (Morrison and Harrington 1979, Senner and Howe 1984, Harrington et al. 1991). Growing evidence (Schneider and Harrington 1981) indicates that staging areas are unusually productive sites with highly predictable but seasonally ephemeral "blooms" of invertebrates, which shorebirds use for fattening. In some cases, especially for "obligate" coastal species, specific sites are traditionally used; even other species sites may shift between years. Because of this, conservationists believe some species are at risk through loss of strategic migration sites (Meyers et al. 1987). Other species are threatened by the loss of breeding and wintering habitats (Page et al. 1991, Haig and Plissner 1993).

The predicted consequences of global warming, such as sea-level change, will also strongly affect the intertidal marine habitats, which many species of shorebirds depend upon. Some of the strongest warming effects will be at high latitudes, including those where many shorebirds migrate to breed, as well as south temperate latitudes, where many of them winter.

Population trend evaluation has been conducted for 27 of 41 shorebird species common in the United States east of the 105th meridian. Of the 27 species for which trend data are available, 12 show no change, 1 increased, and 14 decreased. There were no clear correlations with habitat.

It is important that shorebird populations are monitored nationally, yet most species are hard to monitor because they inhabit regions that are difficult to access for much of the year. Migration seasons appear to be the most practical time for monitoring most species. Unfortunately, sampling for population monitoring during nonbreeding seasons presents a group of unresolved analytical challenges. Additional work on existing data can help identify how or whether broad, voluntary, or professional networks can collect data that will better meet requirements for monitoring population change.

Piping Plovers – Modified from Haig and Plissner (1995)

The piping plover (*Charadrius melodus*) is a wide-ranging, beach-nesting shorebird whose population viability continues to decline as a result of habitat loss from development and other human disturbance (Haig 1992). In 1985 the species was listed as endangered in the Great Lakes Basin and Canada and threatened in the northern Great Plains and along the U.S. Atlantic coast. The U.S. Fish and Wildlife Service is proposing that birds in the northern Great Plains also be listed as endangered.

Avian and mammalian predation is a problem throughout the species' breeding range, although population numbers appear to be stabilizing on the Atlantic coast and the Great Lakes as a result of using predator exclosures over nests (Rimmer and Deblinger 1990, Mayer and Ryan 1991, Melvin et al. 1992). Human disturbance continues to be a problem on the Atlantic coast (Strauss 1990).

The discovery of the high proportion of wintering piping plovers on algal and sand flats has significant implications for future habitat protection. Current development of these areas on Laguna Madre in Texas and Mexico, increased dredging operations, and the continuous threat of oil spills in the Gulf of Mexico will result in serious loss of piping plover wintering habitat.

In summary, piping plovers suffer from many factors that may cause their extinction in the next 50 years. Most devastated are the Great Lakes and northern Great Plains birds whose viability is severely threatened.

Unfortunately, recovery is hindered by a lack of knowledge about the winter distribution, status of winter sites, adequate water-management policy in western breeding sites, and direct human disturbance on the Atlantic coast.

Aquatic Communities

Fishes – Modified from Walsh et al. (1995)
The Southeast has about 485 known species of native freshwater fishes, representing 27 families. Most of the diversity of the southeastern fish fauna is in five families: the darters and perches (family Percidae; 31.3%); the minnows (family Cyprinidae; 29.7%); the madtoms and bullhead catfishes (family Ictaluridae; 6.8%); the suckers (family Catostomidae; 6.6%); and the sunfishes and basses (family Centrarchidae; 5.8%). The greatest diversity is in the Appalachian Mountains and Interior Plateau, but other regions of the Southeast also harbor many more species than do similar-sized geographic areas elsewhere in the United States.

In the Southeast, fish declines are the result of the same factors that cause global deterioration of aquatic resources, primarily habitat loss and degraded environmental conditions. The principal causes of freshwater fish imperilment in the Southeast and other areas of the United States are dams and channelization of large rivers, urbanization, agriculture, deforestation, erosion, pollution, introduced species, and the cumulative effects of all these factors (Moyle and Leidy 1992, Warren, Jr. and Burr 1994). The most insidious threat to southeastern fishes is sedimentation and siltation resulting from poor land-use patterns that eliminate suitable habitat required by many bottom-dwelling species. Cumulative effects of physical habitat modifications have caused widespread fragmentation of many fish populations in the Southeast, presenting difficult challenges for those trying to reverse and restore diminished fish stocks.

Aquatic resources are often resilient and capable of recovery, given favorable conditions. Conservation of southeastern fishes will require significant changes in land management and socioeconomic factors (Moyle and Leidy 1992, Warren, Jr. and Burr 1994), but such changes are necessary to stem future losses of biodiversity. The first step required is to improve public education on the value and status of native aquatic organisms. For resource managers and policy makers, increased efforts must be made to assume proactive management of entire watersheds and ecosystems; establish networks of aquatic preserves; restore degraded habitats; establish long-term research, inventory, and monitoring programs on fishes; and adopt improved environmental ethics concerning aquatic ecosystems (Warren, Jr. and Burr 1994). The southeastern fish fauna is a national treasure of biodiversity that is imminently threatened. If this precious heritage is to be passed on, its stewardship must be improved through cooperative actions of all public and private sectors within the region.

Freshwater Mussels – modified from White et al. (1998)
The Southeast's freshwater mussels include 270 species and subspecies in 49 genera, representing 90% of the freshwater mussel fauna of all of North America north of Mexico (Williams et al. 1993). Ten genera are endemic to the Southeast. Of 93 species and subspecies limited in the United States to one or two states, 91 occur only in the Southeast. The species richness of freshwater mussels in the Southeast is attributed to habitat diversity (including substrates of attachment), evolution within isolated river basins, stream capture over geologic time (which produces new patterns of dispersal and isolation), and high richness in fish species (larval forms use fish as hosts).

Forty-eight percent of the freshwater mussels of the Southeast are endangered, threatened, or possibly extinct (Williams et al. 1993, Williams and Neves 1995). An additional 25% are of special concern, resulting in 73% of this diverse fauna being at risk. Only 25% of the fauna is considered stable (Williams et al. 1993). Of 21 species that are now potentially extinct, 14 were endemic to the Southeast (Williams et al. 1993). Declines in freshwater snails and other mollusk groups are probably also occurring in the Southeast, but surveys of these groups are less complete.

Declines in mussel faunas have affected river basins region wide, including those with higher and lower amounts of endemism. Historically, diversity of mussels increased from headwaters to the mouths of rivers; pollution and other human influences also increase in this direction. Hence, declines in diversity have been most significant in the lower reaches of rivers. Habitat specialists (those requiring, for example, a particular kind of hard substrate) have declined more than habitat generalists.

Factors that are important in declines in mussel richness and abundance are sedimentation, pollution, changes in river flow due to dams and channelization, invasions of nonindigenous species (for example, the zebra mussel and Asian clam), and loss of fish hosts. In addition, commercial harvest of mussels is causing unknown effects on target and non-target species (Williams et al. 1993). As with other aspects of aquatic diversity, retention of natural vegetation in floodplains and along riverbanks is a key element in the protection of water quality and mussel populations. Many southeastern states still have areas with high mussel diversity and abundance, such as the Clinch River in Virginia, Swift Creek in North Carolina, Stephens Creek in South Carolina, and the Ogeechee River in Georgia. These waters tend to be tributary and headwater rivers within drainage basins of several hundred square kilometers in which silviculture is the dominant land use and agricultural and urban areas are limited.

No region wide monitoring or conservation plan exists for freshwater mussels. Conservation efforts will require cooperation of many public and private groups because mussel populations ultimately depend on water quality that is affected by human activities over large areas. The growing human population and its need for sources of clean drinking water will increase the pressure for the creation of additional reservoirs, which in turn will further imperil this distinctive element of the southeastern fauna.

Species with Conservation Status

Based on uncertified data, fifty nine vertebrate species with designated federal conservation status (i.e., Threatened, Endangered, or Candidate) are known to be present in Southeast Coast Network Parks (See Appendix 10). A total of 206 species with State listing status have also been identified within Network parks. In both cases the number is likely *under*estimated and will be revised as baseline inventories of vertebrates and vascular plants are completed during the next three years. Species of conservation status that are found on more than two parks include Red Cockaded Woodpecker, gopher tortoise, eastern indigo snake, piping plover and other shorebirds, wood stork, bald eagle, and all sea turtle species. Seven cetacean species, all of which are listed as “Endangered,” have also been documented at coastal parks within the network but management of those species is limited to beaching events rather than of overall species conservation. Declines of the majority of these species are related to changes in available habitat, primarily in longleaf pine forest and coastal dune ecosystems. Agents of change include fire suppression, visitor use, and adjacent or historical land uses (primarily residential and forestry).

Landscapes

Corridors

Corridors are habitats that join two or more “patches” within the landscape, and are a widely-accepted and popular management tool in conservation biology to facilitate movement of organisms and gene flow among populations that might be otherwise isolated within the landscape (Lewis, Jr. 1964, Puth and Wilson 2001). Although often used, little research has been done to study whether corridors actually meet their design goals. Recent studies have shown that although corridors can increase the exchange of organisms between patches, the rate of that exchange is largely dependent on factors such as corridor design / habitat characteristics, natural migration rates, population growth rates, and home ranges of target species (Hudgens and Haddad 2003, McDonald and St.Clair 2004). Because these factors vary among species, the degree to which any given corridor is used is largely species dependent, and can have differing benefits over both the short and long term. In general, though, corridors are used by broad suites of species with varying life histories and can facilitate key ecosystem function (such as seed dispersal and pollination) in addition to migration (Tewksbury et al. 2002, Haddad et al. 2003).

Several parks within the Network contain significant habitat corridors within the larger landscape, particularly where local green space initiatives are acquiring land to connect lands along riparian river corridors (i.e., CHAT, KEMO, and OCMU). The EPA’s Southeastern Ecological Framework provides guidance to regional planners for strategically protecting lands to increase connectivity among natural areas by purchasing or creating easements that can act as corridors (Carr et al. 2002).

Agents of Change

Individuals & Populations

Wildlife Disease – modified from U.S. Geological Survey (2004)
Disease has long been recognized as one of the potentially limiting factors on wildlife populations. Now, the rapid spread of established diseases; the emergence of new diseases in humans, domestic animals, and wildlife; and the threats of bioterrorist attacks have attracted considerable public attention, as well as generated a call for action. In addition, convincing evidence has been presented advocating the usefulness of wildlife as sentinels for public and domestic animal health threats. Emerging zoonotic diseases (transmissible between animals and humans) have been identified as significant public health threats.

Further, animal disease can be included as one of the threats to global biodiversity. International trade in animal and plant species (including invasive pathogens), human population increases, and reduced wildlife habitat all create situations in which disease outbreaks may occur with increased frequency. Additionally, intermingling of livestock and wildlife creates new opportunities for disease transmission. Recent outbreaks of Chronic Wasting Disease (CWD), West Nile Virus, and House Finch Mycoplasmosis are notable examples of diseases of concern.

Species, Assemblages, & Communities

Exotic / Invasive Species – modified from Williams and Meffe (1998)
Invasion by nonindigenous species is one of the most important issues in natural resource management and conservation biology today. The ability of nonindigenous species to alter population, community, and ecosystem structure and function is well documented (Elton 1958, Mooney and Drake 1986, Vitousek et al. 1987, Drake et al. Drake et al. 1989). Ecosystem-level changes that alter water, nutrient, and energy cycles; productivity; and biomass directly affect human society. Ecosystem-level consequences of invasive nonindigenous species have major ecological and economic implications and directly affect human health. Complex technology has addressed the cleanup of chemical pollutants and contaminants and has reversed some of the damage from physical alteration of the environment. However, little attention has been paid—and almost no progress has been made—In addressing the problem of nonindigenous species.

The problem of biological invasion of the United States is not new. In the continental United States, it began with the arrival of the first European settlers more than 500 years ago and has continued at an increasing rate. In Hawaii, it began more than a thousand years ago with the arrival of the Polynesians, who introduced several plants into their new landscape. Many of the early introductions of plants and animals were intentional and generally viewed as a welcome enrichment of the American biota. Among early introductions were the domesticated animals and plants, which were essential to the survival of settlers as dependable sources of food and fiber. As invasive nonindigenous species have increased and their effects on native biota have become apparent, the perception about many introductions has shifted from welcome additions to pests. Today, although the economic and recreational benefits of selected nonindigenous species are considerable, evaluation of the economic and ecological costs reveals that introductions of nonindigenous species can also be expensive. The nonindigenous species problem has reached proportions that demand development of a coherent national policy to guide future actions.

Definitions of invasive nonindigenous species have been inconsistent, leading to confusion in lay and scientific literatures. First, the distinction between natural biological invasions, which are generally considered as range expansions, and introductions involving human activities is important. Exotic, alien, transplanted, introduced, nonindigenous, and invasive are words that have been used to describe plants and animals that were moved beyond their native ranges by humans. For consistency, we adopted the definition from the Nonindigenous Aquatic Nuisance Prevention and Control Act of 1990 (Public Law 101-646, 16 USC 4701-4741, approved 29 November 1990), which defines nonindigenous species as, “The condition of a species being moved beyond its natural range or natural zone of potential dispersal, including all domesticated and feral species and hybrids.” This definition embodies the most critical biological aspect of nonindigenous species—the movement or introduction of a species beyond its native range by humans. Some resource managers have used political boundaries, such as state or national borders, as a standard to determine the status of an introduction (Shafland 1986); however, they did not consider that species moved beyond their native ranges within state boundaries or within the United States (for example, from the east coast to the west coast) as nonindigenous species. Regardless, ecosystems receiving nonindigenous introductions respond based on a suite of biological and ecological interactions irrespective of the political boundaries from which the species originated. Today, many biologists recognize that any species moved beyond its native range by human activities is a nonindigenous species, and they realize that when such an action is taken, it is hazardous to the economic and ecological foundations of the country (Office of Technology Assessment, United States Congress Office of Technology Assessment, United States Congress 1993).

An examination of the origin of nonindigenous species introductions helps us understand the problem. Nonindigenous species can be divided into three categories: intentional introductions, intentional introductions with subsequent escape, and unintentional introductions. Intentional introductions are those nonindigenous species transported beyond their native range and released into the wild with the purpose and intention that they will become established; these include the house sparrow and the common carp. Intentional introductions with subsequent escape are those nonindigenous species transported beyond their native range under captive conditions and which subsequently escape into the wild, where they may establish reproducing populations; these include aquarium fishes and the African clawed frog. Unintentional introductions are those nonindigenous species that are transported, usually unnoticed or without detection, beyond their native range in the course of some unrelated activity; these include zebra mussels in ballast water or imported red fire ants on cattle boats.

Nonindigenous species are a major threat to endangered and threatened biota. Invasive nonindigenous plants and animals should be treated as biological pollutants that can, in the presence of physical habitat alteration or chemical pollution, push native biota to or past the brink of extinction. In some environments, the association of nonindigenous species and disturbed habitat makes the evaluation of the effects of invaders on threatened biota difficult to assess. There is no question, though, that invasive nonindigenous species represent an additional stress to threatened biota.

Brown (1989) presented five rules of biological invasions that are broad generalizations about the conditions under which nonindigenous invasions may succeed. These rules, he believed, are broadly applicable to vertebrates, invertebrates, and plants. Of vital importance among these rules is the ecological isolation of the invaded habitat, which seems to be critical to its vulnerability to invasion by nonindigenous species. It also helps if the physical characteristics of the new environment are similar to those in the native environment of the invasive nonindigenous species and if other species are not present in similar niches of the invaded habitat. Finally, disturbance and close association with humans are common characteristics of a community that is vulnerable to invasion by nonindigenous organisms.

Nonindigenous species do contribute to a significant proportion of listings of threatened and endangered species in the United States; about 315 native species and subspecies listings are attributed, at least in part, to nonindigenous species. Based on U.S. Fish and Wildlife Service Federal Register listing documents for endangered and threatened species, nonindigenous species have contributed to the decline of approximately 35% of listed taxa. One-third of the 35% of the listed taxa affected by nonindigenous species are from island ecosystems, primarily Hawaii and Puerto Rico. Although island taxa seem to be more susceptible to the adverse effects of nonindigenous species, the mainland biota is far from immune.

Of the approximately 300 freshwater mussels found in the United States, about 73% are considered imperiled (Williams et al. 1993). Scientists believe that two nonindigenous bivalve mollusks, Asian clams and zebra mussels, have contributed to the decline of native mussels (Ricciardi et al. 1995). Asian clams, the most widespread nonindigenous mollusks in the United States, entered the west coast in the 1930's and invaded the southeastern United States in the 1950's (McMahon 1983). In some areas, Asian clams carpet stream bottoms, reaching densities of several thousand individuals per square meter.

Feral Cats – modified from Queensland Department of Natural Resources, Mines and Energy (2003)
A descendant of the African wild cat (*Felis silvestris lybica*), the common 'house' cat (*Felis catus*) has now been domesticated for about 4,000 years. Although the domestic cat has a long history of association with man, it retains a strong hunting instinct and can easily revert to a wild (feral) state when abandoned or having strayed from a domestic situation.

Semi-feral cats live around dump sites, alleys or abandoned buildings, relying on humans by scavenging rubbish scraps and sheltering in abandoned structures. The true feral cat does not rely on humans at all, obtaining its food and shelter from the natural environment.

The feral cat differs little in appearance from its domestic counterpart; however, when in good condition, the feral cat displays increased overall muscle development, which is especially noticeable around the head, neck and shoulders, thus giving the animal a more robust appearance. The average body weight of male feral cats is from 3 kg to 6 kg, while that of females varies from 2 kg to 4 kg. Body weights vary with condition, with some extremely large specimens having been documented.

The feral cat is most active at night, with peak hunting activity occurring soon after sunset and in the early hours before sunrise. At night the cat displays a distinctive green eyeshine under spotlight, making it easily distinguishable from other animals. During the day it will rest in any number of den sites, which may include hollow logs, dense clumps of grass, piles of debris, rabbit burrows, and even the hollow limbs of standing trees.

Male cats attain sexual maturity at about 12 months of age, whereas females are capable of reproduction at approximately 7 months. Annually, and under ideal conditions, an adult female can produce up to three litters—each of usually four kittens, but varying from two to seven.

As the breeding instinct is triggered by the increasing length of daylight, litters are less frequent in winter. Most reproduction occurs during the spring and summer months, and is generally limited to two litters per year. Birth follows a gestation period of 65 days, and kittens may be reared in a single den site or may be frequently shifted to other sites within the female's home range. Family and litter bonding begin to break down when the kittens are approximately seven-months old. The female's ability to bear litters does not decrease with age, so reproduction continues for the course of her life.

The energy expended by an adult male cat requires it to consume 5% to 8% of its body weight in prey per day, while females raising kittens require 20%. Based on these figures, one study concluded that 375 feral cats on Macquarie Island would consume 56,000 rabbits and 58,000 sea-birds per year. Where present on the mainland, rabbits may comprise up to 40% of the diet of feral cats. Cats are successful as a control mechanism only when rabbit densities are low. At other times cat predation does little to halt the build-up or spread of rabbit populations; rabbits merely help to support a larger number of cats. When seasonal shortages of rabbits occur there is a corresponding rise in the number of native animals taken by cats.

The feral cat is an opportunistic predator, and dietary studies have shown that small mammals, birds, reptiles, amphibians, insects and even fish can be taken as prey. Cat predation is particularly harmful in island situations, and a number of species have become extinct due to the introduction of cats by early sealers and lighthouse keepers. On the mainland, native animals—which already suffer due to the destruction of their habitats by man and other introduced animals—may be endangered further by cat predation. Actual competition for prey can cause a decline in the numbers of native predatory species such as, eagles, hawks and reptiles.

Forest Pest Species

Southern Pine Beetle – modified from Meeker et al. (1995)
The southern pine beetle (SPB), *Dendroctonus frontalis*, is the most destructive insect pest of pine in the southern United States. A recent historical review estimated that SPB caused $900 million of damage to pine forests from 1960 through 1990 (Price et al. Price et al. 1992). This aggressive tree killer is a native insect that lives predominantly in the inner bark of pine trees. Trees attacked by SPB often exhibit hundreds of resin masses (i.e., pitch tubes) on the outer tree bark. SPB feed on phloem tissue where they construct winding S-shaped or serpentine

galleries. The galleries created by both the adult beetles and their offspring can effectively girdle a tree, causing its death. SPB also carry, and introduce into trees, blue-stain fungi. These fungi colonize xylem tissue and block water flow within the tree, also causing tree mortality (Thatcher and Conner 1985). Consequently, once SPB have successfully colonized a tree, the tree cannot survive, regardless of control measures. When beetle populations are low (endemic), attacks are generally restricted to senescent, stressed or damaged pines; however, epidemics periodically occur (Thatcher et al. Thatcher et al. 1980). During epidemics, SPB infestations often begin in weakened or injured trees, but the high beetle populations can invade and overcome healthy vigorous trees by attacking in large numbers over a short period of time (Thatcher et al. Thatcher et al. 1980). Widespread and severe tree mortality can occur during epidemics. SPB spots (groups of infested trees) may expand at rates up to 50 ft. (15 m)/day, and uncontrolled infestations may grow to thousands of acres in size (Ron Billings, Texas Forest Service, personal communication). SPB attacks are not limited to conventional forest sites; they also may kill high-value trees in yards, parks, and other ornamental settings (Thatcher et al. 1978). Because of the seriousness of SPB infestations, care should be taken not to confuse SPB with the less aggressive but more common pine bark beetles of Florida, the pine engravers (*Ips* spp.) and the black turpentine beetle (*D. terebrans*) (Dixon 1984, Dixon 1986).

Outbreaks of this insect tend to be cyclical in occurrence. Outbreaks have occurred on 6-12 year intervals and generally last for 2-3 years in areas were SPB has long been a problem. Throughout the South, the time between outbreaks has decreased while the intensity and distribution of each outbreak has increased since 1960 (Price et al. Price et al. 1992, Belanger et al. 1993). In Florida, infestations have been relatively few and small in the past (Chellman and Wilkinson 1975, Chellman and Wilkinson 1980). Many factors are involved in the development of outbreak conditions, such as the abundance and susceptibility of preferred hosts, and weather patterns and events (*e.g.*, drought, storms). Historically, Florida has not experienced many destructive SPB episodes probably because of the lack of large contiguous areas of loblolly and shortleaf pine in susceptible stages. However, an epidemic in and around Gainesville in Alachua Co. during 1994, warrants reconsideration of the serious threat SPB poses to Florida's pine forests. Forest inventory statistics indicate that over the last 25 years, the acreage of loblolly pine forest in Florida has more than doubled from a mere 337,000 ac. (136,380 ha) to more than 750,000 ac. (303,515 ha) (Knight 1969, McCluer 1970, Brown 1987, Clark, Jr. and Sheffield 1994). The current acreage of loblolly pine also represents an all-time high since inventory statistics were reported in 1949. This alarming increase and current level of preferred host material suggests that SPB epidemics in Florida may be more frequent, widespread and destructive in the future.

Aquaculture & Fisheries

Invertebrate Fisheries – modified from Nance and Harper (1999)

Important recreational and commercial marine invertebrates in the southeastern United States include shrimp, spiny lobster, stone crab, and conch. Some fisheries, as for coral, are almost nonexistent. Others, like the penaeid shrimp fishery, are both extensive and extremely valuable. The southeast region's shrimp fisheries are one of the most valuable U.S. fisheries based on ex-vessel revenue. Some fisheries, such as those for spiny lobster and stone crab, have only moderate value on a national basis but are important locally or regionally. Because of the diversity in species, fisheries, geographic locations, yields, values, etc., each species group in the marine invertebrates unit must be examined separately for proper perspective.

Penaeid shrimp have been fished commercially since the late 1800's. The first fishery used long seines in shallow waters, until the otter trawl, introduced in 1915, extended shrimping to deeper waters. At first, most vessels towed one large trawl, sometimes 120 feet wide at the mouth. Soon, a two-trawl arrangement (each about 40–75 feet wide at the mouth) was found more effective. Some shrimpers are using a twin-trawl system which tows four trawls of about 40 feet wide at the mouth. The twin-trawl system is now very common gear on commercial offshore shrimpers.

In the South Atlantic, white shrimp stocks are centered off the Georgia and South Carolina coasts. Brown shrimp are centered off the North and South Carolina coasts. The Atlantic fishery is much smaller than that of the Gulf and currently is managed under a Federal fishery management plan implemented in November 1993. This provides for compatible state and Federal closures if needed to protect overwintering shrimp stocks. A subsequent amendment added rock shrimp to the fishery management plan.

Brown, white, and pink shrimp account for 90% of the total Gulf of Mexico shrimp catch. In 1997 alone, these three important species produced 84,967 t valued at over $437 million in ex-vessel revenue. They are found in all U.S. Gulf waters inside 120 m depths. Most of the offshore brown shrimp catch is taken at 20-40 m depths, white shrimp are caught in 10 m or less, and pink shrimp in 20-30 m. Brown shrimp are most abundant off the Texas-Louisiana coast, and the greatest concentration of pink shrimp is off southwestern Florida. In the South Atlantic, white shrimp landings are about 21% of their Gulf counterparts, while brown and pink shrimp are around 6% of the Gulf yield. Current, recent, and long-term potential yields for these species are given in.

The shrimp fisheries are currently overcapitalized, with more fishing effort being expended than needed to harvest the resource. In addition, the harvesting of small shrimp inshore is sacrificing yield and value of the catch by cutting short future growth.

Shrimp fisheries use small-mesh nets and can catch non-target species such as red snappers, croakers, seatrouts, and sea turtles. Juvenile finfish are often harvested, and this may be a major source of mortality for them. Some fish caught by shrimpers are currently at low stock levels. This bycatch may slow or prevent recovery if not mitigated.

As sea turtles are all listed as endangered or threatened under the Endangered Species Act, shrimp vessels have been required to use turtle excluder devices in their nets since 1988 to avoid capturing sea turtles and thus protect the stocks.

Table A7-1. Productivity in metric tons and status of Caribbean invertebrate fisheries (Nance and Harper 1999).

Species and area	Recent average yield (RAY)[1]	Current potential yield (CPY)	Long-term potential yield (LTPY)	Fishery utilization level	Stock level relative to LTPY
Brown shrimp, Gulf of Mexico	53,080	Unknown	57,653[2]	Full	Near
Brown shrimp, Atlantic	2,645	Unknown	3,447[2]	Full	Near
White shrimp, Gulf of Mexico	28,942	Unknown	29,980[2]	Full	Near
White shrimp, Atlantic	6,045	Unknown	6,305[2]	Full	Near
Pink shrimp, Gulf of Mexico	11,009	Unknown	7,469[2]	Full	Near
Pink shrimp, Atlantic	730	Unknown	955[2]	Full	Near
Royal Red Shrimp	250	Unknown	Unknown	Unknown	Unknown
Seabob Shrimp	3,947	Unknown	Unknown	Unknown	Unknown
Rock Shrimp	6,240	Unknown	Unknown	Unknown	Unknown
Spiny lobster, SE United States[3]	3,325	2,400	3,565	Over	Below
Spiny lobster, Caribbean	111	Unknown	Unknown	Unknown	Unknown
Stone crab[4]	2,961	1,121	976	Full	Near
Queen conchs[5]	91	55	Unknown	Over	Below
Coral[6]	0	0	Unknown	Unknown	Unknown
Total	119,376	116,575	120,953		

[1]1995–97 average for shrimp; 1994–96 average for other species.

[2]Long-term potential of brown, white, and pink shrimp based upon last observed 10-year average annual yield (1988–97).

[3]Yields based upon commercial catches; recreational catch is unknown but may be significant.

[4]Yields are in tons of claws; declawed crabs regenerate new claws.

[5]Landings from Puerto Rico. Fishing prohibited in Florida.

[6]Coral harvests prohibited except for a small take allowed for use in aquarium and pharmaceutical industries.

Other Nearshore Fisheries – modified from Anderson et al. (1999)
In the southeast as in the northeast, the recent average yields reported here are underestimated, because they can generally be based only on commercial landings. Recreational landings, which may be considerable, are generally unavailable for the invertebrates that dominate the southeast nearshore fisheries. Bycatch mortality is not estimated, or is incompletely estimated, for many species.

Blue crabs dominate the nearshore catch by weight. Recent landings have fluctuated around 60,000 tons (Table A7-1. Productivity in metric tons and status of Caribbean invertebrate fisheries (Nance and Harper 1999).). Oyster harvests have trended downward over the last decade, but recent landings have been steady with a recent average yield of 10,440 tons. Calico scallop has been important in the landings in the past (20,000 t in 1984), but recent landings have averaged 1,184 tons.

Mullet landings in the region have been affected by a ban on nets over 500 square feet in Florida's waters. This ban took effect July 1, 1995. Recent average yield is down to 12,558 t, but more telling are the landings for 1996, which were reported at 9,484 t. Commercial landings outweigh the recreational catch by slightly more than 10:1. Herrings (not including American shad, alewife, or blueback herring) and Spanish sardine recent average yields total 6,040 t in the southeast, almost all from commercial landings. Bait fisheries for species such as ballyhoo and bigeye scad (goggle-eye) exist primarily in south Florida, with a net fishery for bigeye scad in the Florida panhandle. A major portion of the bigeye scad were landed in Palm Beach County prior to the state-issued net ban. Present landings in that area are a result of a live-bait fishery and have a high value. Ballyhoo landings from the Palm Beach area also dropped, but those in the Florida Keys have been steady. Flying fish are often landed with the ballyhoo.

Table A7-2. Productivity in metric tons and status of southeast nearshore fishery resources (Anderson et al. 1999).

Species	Recent average yield (RAY[1])	Commercial RAY[1]	Recreational RAY[1]	Fishery utilization level	Stock level relative to LTPY[2]
Blue crab	59,876	59,876	Unknown	Unknown	Unknown
Mullet	12,558	11,537	1,021	Unknown	Unknown
Oysters	10,440	10,440	Unknown	Unknown	Unknown
Other herring & Spanish sardine	6,040	5,980	60	Unknown	Unknown
Flounder (southern & Gulf)	1,514	609	905	Unknown	Unknown
Bait shrimp	1,264	1,264	Unknown	Unknown	Unknown
Calico scallops	1,184	1,184	Unknown	Unknown	Unknown
Ballyhoo, bigeye scad, flyingfish	664	644	20	Unknown	Unknown
Total	93,540	91,534	Unavailable		

[1]RAYs are for 1994–1996. Recreational data are estimates based on surveys.
[2]LTPY = Long-term potential yield.

Mosquito Control – modified from (U.S. Environmental Protection Agency 2002b)
The first step in mosquito control is surveillance. Mosquito specialists conduct surveillance for diseases harbored by domestic and nonnative birds, including sentinel chickens (used as virus transmission indicators), and mosquitoes. Surveillance for larval habitats is conducted by using maps and aerial photographs, and by evaluating

larval populations. Other techniques include various light traps, biting counts, and analysis of reports from the public. Mosquito control programs also put high priority on trying to prevent a large population of adult mosquitoes from developing so that additional controls may not be necessary. Because mosquitoes must have water to breed, methods of prevention include controlling water levels in lakes, marshes, ditches, or other mosquito breeding sites, eliminating small breeding sites if possible, and stocking bodies of water with fish species that feed on larvae. Both chemical and biological measures may be employed to kill immature mosquitoes during larval stages. *Larvicides* target larvae in the breeding habitat before they can mature into adult mosquitoes and disperse. Larvicides include the bacterial insecticides *Bacillus thuringiensis israelensis* and *Bacillus sphaericus*, the insect growth inhibitor methoprene, and the organophosphate insecticide temephos. Mineral oils and other materials form a thin film on the surface of the water that causes larvae and pupae to drown. Liquid larvicide products are applied directly to water using backpack sprayers and truck or aircraft-mounted sprayers. Tablet, pellet, granular, and briquet formulations of larvicides are also applied by mosquito controllers to breeding areas.

Adult mosquito control may be undertaken to combat an outbreak of mosquito-borne disease or a very heavy nuisance infestation of mosquitoes in a community. Pesticides registered for this use are *adulticides* and are applied either by aircraft or on the ground employing truck-mounted sprayers. State and local agencies commonly use the organophosphate insecticides malathion and naled and the synthetic pyrethroid insecticides permethrin, resmethrin, and sumithrin for adult mosquito control.

Mosquito adulticides are applied as ultra-low volume (ULV) sprays. ULV sprayers dispense very fine aerosol droplets that stay aloft and kill flying mosquitoes on contact. ULV applications involve small quantities of pesticide active ingredient in relation to the size of the area treated, typically less than 3 ounces per acre, which minimizes exposure and risks to people and the environment.

Habitats & Ecosystems

Visitor Use Impacts

Visitor uses of natural resources, though appropriate within the Natural Park System, can cause a number of direct and indirect changes within the ecosystem. Use of natural areas has increased steadily since 1965 (Cole 1996) and will likely continue as protected lands become increasingly rare in the rapidly developing landscape within the Southeast. Most visitor use impacts are assumed to increase as visitation rates increase.

Of great concern within the SECN are the effects of off-road and recreational vehicles. Several parks allow access vehicular access (such as Cape Hatteras NS), and are potentially affected by increased mortality of sensitive species (sea turtles, ground-nesting shore birds). Also, vehicles can potentially cause changes in vegetation stability, and increased soil erodibility (Grantham et al. 2001). These impacts are potentially of highest concern in areas with loosely-consolidated soils such as coastal dunes. Additional visitor use impacts include degraded air quality (associated with vehicle and off-road vehicle use), litter, trampling, poaching, species introductions, and social trail creation. Also, visitor use impacts can cause changes in animal ranges, particularly large mammals through avoidance behaviors (Papouchis et al. 2001). Human-animal interactions might be of high importance in wilderness areas during breeding times. Also, trails have been shown to be a vector for dispersion of exotic plant species (Patel and Rapport 2000).

Landscapes

Natural Disturbances – modified from White et al. (1998)

The Southeast's frequent thunderstorms provide an ignition source for natural fires. In the past, Native Americans and European settlers also burned natural vegetation regularly. Regardless of ignition source, fire frequency and intensity have been dominant forces throughout the Southeast on all but the wettest and coldest (high mountain) sites. The mid- to late 1900's represent a period of reduced fire frequency, size, and intensity, a shift that is a major source of change in the region's ecosystems, leading to increases in mesic species (that is, species adapted to moister conditions), increased understory stem density, increased woody cover in formerly open habitats, and decreases in fire-dependent species and ecosystems.

Tropical storms are also a major recurrent disturbance, with much of the area experiencing about two damaging storms per decade. Between 1871 and 1981, 138 tropical storms affected south Florida (Davis and Ogden 1994). Although storm incidence declines from coastlines to the interior, tornadoes are more frequent in interior areas, where nearly 10 violent tornadoes per year have occurred over the last 100 years (Grazulis 1984, Martin and Boyce 1993).

The heavy rainfall that accompanies these and less violent storms is an important natural disturbance, especially in the Appalachian Mountains, where debris avalanches create open habitats in the forested matrix and flash floods scour stream banks and affect stream biota. Throughout the Southeast, the natural flooding and erosional dynamics of rivers were and are an important natural process for biological diversity; impoundments, changes in the quality and quantity of water, draining of bottomlands, and channelization of rivers are major causes of loss in the biological diversity dependent on dynamic stream and river systems.

Fire / Fire Suppression – modified from White et al. (1998)
Fire was and is important to many southeastern ecosystems, including many Coastal Plain and south Florida ecosystems, pine-dominated forests of the Coastal Plain and Appalachian Highlands, oak and oak-hickory forests, oak savannas, glades, barrens, and prairies. Because most natural communities in the Southeast are dependent on fire, more than 50% of the rarest plants in the region also possess this dependence. Fire may also explain the occurrence of canebrakes, dense stands of the Southeast's only native bamboo, which were frequently described by earlier travelers but which have vanished from the landscape except for small remnant patches (Noss et al. 1995). Although natural fires were quite important, Native Americans and European settlers also set fires frequently. When fire suppression became effective in the 1940's, dramatic changes in ecosystem composition and structure began.

Pine dominance was produced by intense fires, with subsequent lower intensity fires reducing competing hardwoods in the pine understories. Given the age of pine stands, intense, stand-initiating fires must have occurred at least once every 100-200 years; less intense fires occurred much more frequently--every 2-12 years. In the absence of fire, oak, hickory, and pine replace longleaf pine on the Coastal Plain (Stout and Marion 1993), and oak-dominated forests replace pitch pine and Table Mountain pine on the dry ridges of the Appalachians. The net trend of these landscapes is away from pine-dominated ecosystems, leading to declines in species associated with those systems.

Outbreaks of the native southern pine beetle can not only hasten the succession from pine to hardwoods but can also result in high fuel loads. On dry topographic sites and in drought years, high-intensity fires can occur because of these fuel loads. Such hot summer fires are critical to pine regeneration.

Although oaks and hickories increase on the driest sites with reductions in fire, these trees are declining on moister sites where fire was important in limiting mesic hardwoods (Christensen 1977). Thus, throughout the Southeast, there is a general trend toward an expansion of mesic species and a contraction of dry-adapted and fire-dependent species. Understory stem densities have also increased. A failure of oak to regenerate on sites where the species now dominates is a widely observed phenomenon in the eastern United States. McGee (1986) and other researchers hypothesized that this change is caused not only by fire suppression but also by other factors such as air pollution (Kessler, Jr. 1989). Low fire frequencies have also allowed woody plants to invade the glades, barrens, and prairies once associated with oak and hickory forests. Early descriptions of the southeastern landscape suggest frequent forest openings, larger areas of grassland and savanna, and upland forests with open understories (Skeen et al. 1993).

Flooding – modified from White et al. (1998)
The dynamics of flooding and meandering rivers are a major natural process in southeastern ecosystems. Many plant and animal species are dependent on the natural dynamics of water flow. The overall tendency is for human influence to make a dynamic environmental factor less variable. Succession favors the species best adapted to the

more uniform conditions, and diversity decreases. In natural systems, however, extreme hydrological events are an important agent in the maintenance of species diversity.

Changes in Hydrology – modified from White et al. (1998)
Alteration to the hydrological regime is a common disturbance in a variety of southeastern ecosystems: bottomland and floodplain forests, mountain bogs, rocky stream gorges, longleaf pine savanna, Carolina bays, pocosins, Atlantic white-cedar swamps, barrier-island communities, mangrove forests, rivers, streams, caves, lakes, and the Everglades mosaic of communities. Hydrological change has altered flood depth, duration, frequency, and seasonal timing in many of these systems, leading to a raising and lowering of the water table in specific cases.

Hydrological change is caused by sedimentation, construction of dams and other barriers, and channelization (Adams and Hackney 1992). Portions of almost all major southeastern rivers have been impounded during the last 75 years. For example, a 1974 stream survey in Maryland showed that all 14 drainages in 17 tidewater counties had dams (258) or other blockages (89; Lee et al. 1984). Other barriers include farm or mill pond dams, weirs, and raised culverts. Dams result in changes to water temperature and unpredictable releases of water. Channelization, which includes straightening the streambed, smoothing bottom contours, and removing logs, obstructions, and plants, alters the rate and timing of water flow (the local water table is lowered, resulting in increased flooding downstream), aquatic productivity, microhabitats within the channel, and food webs. Sedimentation, blockages, and channelization often occur within one river system, leading to decreases in native fishes and other aquatic species, a loss of species intolerant of such changes, and increases in tolerant species and nonindigenous species (Crumby et al. 1990).

Changes in Water Quality – modified from White et al. (1998)
In recent years, the Clean Water Act has done much to reduce point sources of pollution by requiring water treatment. Nonpoint-source pollution and sedimentation are harder to control, though. Sedimentation is a serious problem for most aquatic organisms, particularly primary producers as well as benthic (bottom-dwelling) macroinvertebrates and fishes that require gravel or rock substrates. Medium-sized rivers are particularly vulnerable to alteration of substrate composition and texture (Etnier and Starnes 1991).

Other factors responsible for depletion of aquatic faunas are pollution (including chemical and thermal pollution) and introduction of non-indigenous fishes and aquatic plants. Invasive non-indigenous plants that are capable of altering function (for example, hydrology, amount of photosynthesis, and food webs) in aquatic systems in the Southeast include parrot feather watermilfoil (*Myriophyllum aquaticum*), Eurasian watermilfoil (*M. spicatum*), waterthyme (*Hydrilla verticulata*), curlyleaf pondweed (*Potamogeton crispus*), water hyacinth (*Eichhornia crassipes*), and water chestnut (*Trapa natans*) (Hotchkiss 1967, Lachner et al. 1970).

Changing Land Use and Watershed Development – modified from White et al. (1998)
The Southeast has one of the country's most rapidly growing human populations. Population growth was 20% from 1970 to 1980, 13.4% from 1980 to 1990, and an estimated 10%-19% for the 1990's (U.S. Bureau of the Census 1994). The continued growth of the human population and changes in the way humans interact with the natural landscape present a challenge to conservationists concerned with the survival of diversity in this biologically rich region.

Destructive logging and soil erosion in the Southeast were major stimuli to the conservation movement in the early twentieth century; this movement led to the creation of national forests, national parks, state parks, research stations, and other protected areas. In contrast to the western United States, the Southeast had little public land--less than 10%--and these areas had to be created by purchase of private lands. Today, public land is mostly in the mountains, with less public land in the Piedmont and Coastal Plain (Boyce and Martin 1993).

Data from 1987 show that although 55% of the land was forested then, the trend was downward, with a decline of 5% since 1960 (U.S. Forest Service 1988, Martin and Boyce 1993). The rest of the land was used for crop and pasture (31%) and miscellaneous purposes (roads, towns, cities, airports; 14%). Urban areas are growing at the fastest rate, but the rate of growth varies by region. For example, in North Carolina, urbanization occurred three times faster in the Piedmont than in either the mountains or Coastal Plain (see review in Boyce and Martin 1993). Although the high total of forested land indicates potential for the survival of biological diversity, these forests are largely privately owned; less than 10% of the forested land is in federal ownership (U.S. Forest Service 1988) and the remainder are not managed for biological diversity per se. Further, because these lands have almost all been disturbed by logging and agriculture, they have already lost communities and species.

Forestland has been predicted to decline by 15% over the next 50 years (with additional forestland converted from natural to plantation forests), agricultural land to decline slightly (with a continued shift from small to large farming operations), and urban areas to increase in area (see discussion in Boyce and Martin 1993), suggesting that further habitat loss and fragmentation will occur near human population centers. We know too little about the survival of biological diversity in human-dominated landscapes, but we do know that the biodiversity of these areas will generally decrease with habitat fragmentation (Harris 1984). Some human-dominated landscapes, however, have the potential to support the diversity of some groups. For example, some crop systems support bird diversity(Allen 1995) by cultivating marginal lands, including some wetlands.

Atmospheric Deposition

Atmospheric deposition is the process by which airborne pollutants are deposited to the earth. These pollutants include, but are not limited to, sulfur dioxide, nitrogen oxides, ammonia, and mercury. Total deposition consists of both wet and dry components.

Wet deposition occurs when pollutants are deposited in combination with precipitation, predominantly by rain and snow, but also by clouds and fog. The NPS monitors wet deposition through the National Atmospheric Deposition Program (NADP). In general, atmospheric deposition is higher in the eastern U.S. due to higher emissions of SO_2, NO_x, and NH_3. Deposition of ammonium is highest through the Midwest and over the Great Plains region, the result of high ammonia emissions associated with agricultural activities, such as fertilizer use and livestock production.

Atmospheric deposition in the U.S. has changed significantly since the 1980s. Sulfate deposition has declined significantly in the East, with the most dramatic decrease in the Northeast. Despite these declines, which resulted from the SO_2 emission reductions required by Title IV of the Clean Air Act, sulfate concentrations in surface waters have not decreased, showing that ecosystems may take several years to recover.

In contrast to the declines in sulfate deposition, nitrate deposition has not changed significantly, although areas of high deposition appear to be shifting west. However, ammonium deposition has increased dramatically in the past 10 to 15 years, affecting the east and the west.

Dry deposition of particles and gases occurs by complex processes such as settling, impaction, and adsorption. Dry deposition is monitored through the Clean Air Status and Trends Network (CASTNet). Total sulfur deposition is much higher in the Eastern U.S. than in the Western states. With few exceptions, wet deposition is the major contributor to total deposition of sulfur. Total deposition of nitrogen is also higher in the Eastern U.S., however higher rates are also estimated for the Rocky Mountains. Again, most sites are dominated by wet deposition, however the majority of nitrogen deposition to Joshua Tree NP and Death Valley NP in southern California occurs as dry deposition. Annual reports for individual park service stations are available from the NPS, and annual network reports are available from the CASTNet web site.

Water Use – modified from Hermann et al. (1998)

From 1950 to 1990, both the population and domestic water use in the United States increased steadily. Withdrawals of fresh and salt waters increased to a peak of 1.7 billion cubic meters per day in 1980, and by 1990 daily freshwater withdrawals were 1.5 billion cubic meters. Rural use of water for households and livestock increased from 1960 to 1990. Irrigation increased from 1950 to 1980, to a maximum of 570 million cubic meters per day, while per capita water use in the United States decreased from 6.8 million cubic meters per day in 1970 to 5.9 million cubic meters per day in 1990. Commercial and industrial uses of water, including self-supplied industrial use and withdrawals of water for mining, increased to a plateau in 1975-1980 before declining by 14%. The estimated use of fresh groundwater—fresh water drawn from below the ground—was 130 million cubic meters per day in 1950. Use of groundwater increased to 310 million cubic meters per day by 1975, decreased during the 1980's to 280 million cubic meters per day, and then increased again to 300 million cubic meters per day in 1990. The use of fresh surface water peaked in 1980 at 1.1 billion cubic meters per day and declined to 980 million cubic meters per day by 1990. Consumptive use—water that is withdrawn from a water source and does not eventually return to the water source—of fresh water followed the same patterns as withdrawals. The reduction of withdrawals during 1980-1985 reflected conservation but could also relate to climate or the economic slowdown (Solley and Pierce 1988, van der Leeden et al. van der Leeden et al. 1990, Solley et al. 1993).

Fresh water is now a limited ecological (physical and biological) and economical resource. The trend in the present use of water reflects its limited availability. Krusé (1969) estimated that by 1965, withdrawals of 1.3 billion cubic meters per day were exceeding the available dependable water supply by 13%. The deficit reflected the need for reusing water, the increased use of salt water, and the lack of new water development opportunities.

Effects of Water Use on Watersheds

The withdrawal of water or the alteration of water quality elicits responses in watersheds—the area drained by a stream or river. These alterations occur even in the most remote places, and responses include changes in biological diversity and ultimately in the entire landscape (Ward and Stanford Ward and Stanford 1979, Becker and Neitzel Becker and Neitzel 1992, Pederson Pederson 1994). In fact, few wild rivers are completely wild, and few native populations are not affected by humans. Benke (1990) etimated that during the past century, 98% of the 5.2 million kilometers of streams in the contiguous 48 states were altered sufficiently by human activities so that they did not meet the more stringent requirements for protection under the Federal Wild and Scenic River provisions. For example, as human population and water use increased, the species diversity of fish communities decreased (Moyle and Leidy 1992). Thus, by 1989, in spite of conservation and restoration, over 100 species of freshwater fishes were added to the threatened or endangered list and more than 250 freshwater fish species were in danger of disappearing (Deacon et al. 1979, Williams et al. 1989, Johnson 1995). The endangerment of freshwater fishes in several regions of the United States has been linked to dams, the straightening of channels of large rivers, the building of cities, the expansion of agriculture, the logging and clearing of headwaters, the erosion of river channels, the pollution of water, and the introduction of nonindigenous species. The total effect of these developments is the alteration of stream ecology as evidenced by changes in the migration patterns of fishes, in stream water temperature and nutrient levels, in water chemistry, and in biological diversity (Warren, Jr. and Burr 1994).

The terrestrial part of the watershed ecosystem is also threatened. Before European settlement, the estimated amount of riparian land in the 100-year floodplains of the lower 48 states was 49 million hectares. By the 1980's it was reduced by 81%, to 9.3 million hectares (Brinson et al. 1981); 22 states had lost more than 50% of their wetlands. Although the rate of change in wetland areas slowed between the mid-1970's and mid-1980's, there was still a net loss, which created a major shift and reduction in the variety of plants and animals in riparian lands (Johnson and McCormick 1979, Petts 1984, U.S. Office of Technology Assessment 1984, Mathias and Moyle 1992).

The total effects of human activities in aquatic and riparian lands are not nearly understood. The change in biological diversity, however, can be linked to habitat change and to the loss of species (Hunt 1988). From alpine and mountain streams to estuaries and deltas, anthropogenic changes have accumulated, and many of the nation's watershed ecosystems have been drastically altered by these changes.

Nature of Water Development and Use

The development of freshwater resources for human use has many consequences for aquatic biota and for riparian and terrestrial species that depend on aquatic ecosystems for food or habitat. Direct human effects include changes in stream and river flows and lake water levels from dams and irrigation (Mesa 1994), the introduction of pollutants (Crowder and Bristow 1988), both intentional and inadvertent introductions of nonindigenous species by providing access pathways (Kitchell 1990, Cloern and Alpine 1991, Mackie 1991) exploitation of selected species, especially fishes and mussels (Hedgecock et al. 1994). Indirect effects on aquatic biota include introductions of extensive atmospheric contaminants (Schindler et al. 1985), widespread use of salts on roads (Likens 1985), change in aquatic species composition from UV-B radiation, change in water nutrient content and temperature from livestock grazing in the riparian zone (Armour et al. 1991), and change in water quality from human development in upstream watersheds (Byron and Goldman 1989, Fisher 1994, O'Dell 1994).

Water developments have single or multiple purposes. For example, stored water may be withdrawn for cooling of electrical power plants, or it may be released for the generation of electric power. A development may provide water for municipal, agricultural, and industrial withdrawals, as well as for recreational uses (boating, fishing, swimming). The American Rivers group (1995) attributed the most frequent threats to the 30 endangered and threatened rivers on their list to dams (13), agricultural (10) and urban (10) runoff, mining (6), and flood-control or navigation demands (6). Other problems include overgrazing, logging, overuse, and sewage. Water projects often must balance competitive uses that can have different direct or indirect effects on aspects of the biological, physical, or chemical environment.

Coastal Barrier Island Evolution / Shoreline Erosion – *Modified from Williams and Johnston (1995)*
Long-term survey data by the U.S. Geological Survey and others, based on analyses of archive maps, reports, and aerial photographs, demonstrate that coastal erosion is affecting each of the 30 coastal states (Williams et al. 1991). About 80% of U.S. coastal barriers are undergoing net long-term erosion at rates of less than 1 m (3.3 ft) to as much as 20 m (65.6 ft) per year. Natural processes such as storms, rise in relative sea level, and sediment starvation (a reduction in volume of sediment transported by rivers reaching the coast), which may also be a result of human interference, are responsible for most of this erosion; but human factors such as mineral extraction, emplacement of hard coastal-engineering structures, and dredging of sand from navigation channels are now recognized as having major effects on shoreline stability.

As the coastal population grows and barriers become urbanized, valuable habitats are being destroyed, and associated negative impacts such as waste disposal, pollution, and changes in freshwater and fine-grained sediment dispersal are altering entire coastal marine and maritime ecosystems. Protecting all remaining undeveloped coastal barriers should be a national priority. Some protection occurs through the Coastal Barrier Resources System, as well as other local, state, and federal programs, including acquisition, restoration, protection, and management programs.

Beach Renourishment – modified from National Oceanic and Atmospheric Administration (2004)
Policy makers recognize the need to manage, protect, and preserve beach environments, many of which have been subject to severe erosion due to storm events, natural longshore and offshore transport, and development. However, it is difficult to establish a standard management strategy or approach to the problem since erosion rates and sand transport patterns are highly variable along the coastline. Current projects rely primarily on "soft stabilization" measures such as beach nourishment and dune stabilization. These techniques reduce the negative "downdrift" effect associated with hard stabilization and create more attractive beach profiles and dune ecosystems that can promote tourism.

Coastal programs and initiatives exist at both the federal and state levels. Programs at the federal level are administered by the Coastal Programs Division of the Office of Ocean and Coastal Resource Management; however, each state within the OPIS study area maintains its own management program to conserve and protect state coastal resources. The Florida Coastal Management Program, Georgia Coastal Management Program, North Carolina Division of Coastal Management, and South Carolina Coastal Programs Division maintain primary responsibility for coordinating these efforts in their respective states. However, beach renourishment projects tend to be quite complex, involving additional federal and state agencies and associated legislation. The sections listed above will address these complexities.

The coastal environment contains many sensitive habitats and species that can be adversely affected when the system is subject to disturbance. A wide range of impacts are possible during the process of either mining offshore sand sources or depositing the sand on an erosional beach. These impacts must be considered within the project planning process to ensure that negative effects are mitigated. Three main categories of impacts will be discussed in the following sections, along with the related laws and regulations.

Dredging projects can disturb critical habitat for fish species or adversely affect bottom dwellers such as crabs, bivalves, lobsters, and other commercially valuable species. In the nearshore area, dredging can also disturb seagrass bed life cycles. In addition to their photosynthetic properties, seagrass beds are known to serve as a protective environment for the larval development of many marine species. A temporary increase in turbidity and sedimentation could negatively impact ambient water quality.

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